Environmental and Efficacy Studies of a Chromated Fluoride Wood Preservative.

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Abstract.

Significant structural failure rates in creosoted electricity distribution poles of Scots pine (***Pinus sylvestris***) are caused by decay of uncreosoted sapwood and heartwood of the interior groundline region. To eradicate the basidiomycetes responsible for this decay a waterborne chromated fluoride preservative paste, Rentex, has been proposed for in-situ remedial groundline injection treatment of creosoted distribution poles. The Rentex formulation has been investigated with a view to firstly establishing its temporal effectiveness in protecting distribution poles from further decay, and secondly, examining its effects on the adjacent environment.

Laboratory toxicity studies established the fluoride concentrations in Scots pine heartwood and sapwood required to provide protection against decay by strains of ***Neolentinus lepideus***, the basidiomycete most commonly associated with internal decay of distribution poles. Field studies of Rentex treated aged pole sections showed that fluoride readily migrated through the cross-section of treated timbers till at twelve to eighteen months after treatment, toxic fluoride concentrations were generally found throughout the groundline region. The results of a microbiological isolation study of 'on-line' poles could not be used to directly corroborate these laboratory and field results as basidiomycetes were infrequently isolated regardless of treatment. However, field pole isolations of the commonly found mould ***Cladosporium resinae*** were significantly reduced for up to sixteen months after remedial treatment and ***C. resinae*** displayed a greater resistance to fluoride than ***N. lepideus*** in laboratory studies.

Field studies of remedially treated creosoted pole sections showed that the chromium component of the preservative, intended to inhibit leaching of fluoride from the timber, did not migrate and was restricted to the sites of preservative injection. At positions remote from the injection sites therefore, fluoride remained mobile. Consequently, at twenty months after remedial treatment, fluoride concentrations within the susceptible groundline area had generally fallen below toxic levels. Chemical analysis of field soils adjacent to creosoted remedially treated pole sections and 'on-line' poles confirmed that falls in timber fluoride concentrations were due to leaching to the surrounding soil. Leaching of chromium to soil was also established, probably facilitated by its remaining at the injection site which provided a path of little resistance to the movement of this element from the timber.

Field studies of the environmental effects of this characteristic soil contamination were impractical due to economic and seasonal constraints. Therefore to facilitate the measurement of physical and biological indicators of any environmental impact associated with the preservative treatment, a novel physical field model was designed to simulate severe field exposure of remedially treated timber. Each of three models constructed consisted of a layered and drained microbiologically active soilbed supporting an aged creosoted Scots pine pole section. In two models the pole sections had received Rentex treatment. Above each model unit was a source of artificial rainfall to encourage leaching conditions and a source of phtosynthetically active radiation for consecutive crops of Perennial ryegrass (***Lolium perenne***) and Rye (***Secale cereale***).

Soil concentrations of fluoride and chromium adjacent to remedially treated timbers within the model units were found to be significantly higher than background levels in the control model unit and similar to those previously found around remedially treated field poles. The drainage waters of the control unit contained fluoride and total chromium concentrations which were frequently significantly lower than those concentrations found in leachates collected adjacent to treated timbers but compatible with values from

uncontaminated field sites. These results indicated that the physical model provided an accurate picture of the effects of remedially treated timber on the immediate physical field environment.

However, the total quantities of fluoride and chromium found in drainage waters from the contaminated model units did not indicate that treated timber in the field would represent a serious contamination risk to groundwater supplies. Similarly, though elevated soil concentrations of fluoride and chromium resulted in reduced dry matter yield and bio-accumulation of both elements in swards of ***L. perenne***, and reduced soil microbial activity (dehydrogenase levels), these effects were restricted to within ten centimetres of the treated timber. No long term phytotoxic effects over larger areas around treated pole sections were indicated by studies of ***S. cereale*** crops. These model results showed that the environmental impact of remedially treated field poles was likely to be minimal.

The physical field model was successful in allowing the accurate measurement of a number of physical and biological indicators of the environmental impact of remedially treated timber. The potential for further development of the physical model for inclusion in other environmental studies of chemicals and chemically treated structures is clear.

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CHAPTER 3. ENVIRONMENTAL STUDIES OF REMEDIALLY TREATED DISTRIBUTION POLES USING A PHYSICAL FIELD MODEL.

CHAPTER 1.

GENERAL INTRODUCTION.

1.1. Wood.

Wood is one of man's most valuable renewable resources (Levy and Dickinson, 1981). Its use in early communities for the manufacture of implements, provision of shelter and as a fuel source (Jane, 1957) is mirrored by similar use today in technically under-developed countries. In developed countries wood is extensively used in paper manufacture and the production of synthetic textiles. In addition, although synthetic alternatives are available, wood is also the preferred material for building construction and the production of high quality furnishings and packaging materials, due to its availability, natural toughness and strength properties allied to its ease of conversion for use and the aesthetic appeal of the finished product. These fundamental qualities are responsible for a major exterior use of wood in the production of electricity distribution poles, which in the United Kingdom are primarily of the softwood Scots pine (***Pinus sylvestris*** Linnaeus).

1.2. Wood: structure and deterioration.

Softwoods possess two distinct zones, namely the outer sapwood and inner heartwood. The sapwood is that portion of the wood which contains living cells and reserve materials such as starch, whereas the heartwood is that portion which has ceased to contain living cells and where nutrient reserves have been removed or converted to heartwood substances (Wilkinson, 1979). Wood in the living tree contains food storage, translocation and structural units (King, 1981). Four basic types of cells are produced in hardwoods but only two are present in softwoods (Dinwoodie, 1989). These are the tracheids and ray or parenchyma cells making up 90-95% and 5-10% of the total volume respectively (Sjostrom, 1981; Dinwoodie, 1989). Tracheids are the major structural elements of softwoods providing physical support for the tree by virtue of having their

long axes in the vertical plane, and allowing the vertical movement of nutrients in the sap (King, 1981). The flow of sap from one tracheid to another is achieved via small pits in the tracheid wall (Wilkinson, 1979). Parenchyma tissue in softwoods is distributed in the horizontal plane, primarily in the medullary rays, and usually has no structural function within the tree (King, 1981). Chemically, all wood structural elements consist of three main polymeric compounds: cellulose, hemicellulose and lignin, which form the walls of wood cells. The wood cells contain nitrogenous materials, starch, sugars, and minerals (Kirk, 1973), which form a utilisable energy source for the tree.

The wood cells also represent an attractive substrate for decay organisms, and though wood is prone to deterioration by fire, and chemical and physical decomposition (Levy and Dickinson, 1981), biological decomposition is the primary cause of deterioration (Cartwright and Findlay, 1958; Scheffer, 1973). Though the heartwood region is frequently more resistant to decay than sapwood due to the lack of readily available nutrients and the presence of toxic extractants such as tannins (Wilkinson, 1979), biological decomposition of wood results in a significant economic loss during growth, conversion, storage and use. Of the three major types of biological decomposition; marine borer damage, insect damage and microbiological degradation, decomposition by micro-organisms represents the most sustained threat to wood resources (Scheffer, 1973).

1.3. Wood decay and decay micro-organisms.

Fungi are the major causal agents of decay and failure of wooden structures and have been categorised into two groups on the basis of their decay effects, namely the staining and mould fungi, and the wood rotting fungi (Cartwright and Findlay, 1958).

Micro-organisms commonly found in distribution poles in the United Kingdom contain members of both groups.

The staining and mould fungi are members of the Ascomycete and Fungi Imperfecti groups and passively invade the parenchyma cells (Corbett, 1963) to utilise the nutrients within. As these fungi do not degrade the structural components, cellulose and lignin, of wood cells (Butcher, 1966) they do not cause significant strength or weight losses in timber. However, King (1981) has suggested that given the correct environmental conditions, these micro-organisms may cause more serious decay.

Wood rotting fungi are classed as either soft rot fungi belonging to the Ascomycete and Fungi Imperfecti groups, or brown/white rot fungi belonging to the Basidiomycete group. The softened wood surfaces, typical of timber decayed by soft rot fungi led Savory (1954) to give these fungi their name. Soft rot fungi produce cavities within the cellulose layers of cell walls by virtue of their ability to secrete cellulases. They require high moisture contents and a ready source of nitrogen. Soft rot in timber is usually confined to the outer few millimetres of exposed wood, a limit apparently imposed by oxygen supply (Savory, 1955). The Basidiomycetes are responsible for the greatest reduction in strength and durability of exposed timber in temperate regions. They have no requirement for high moisture contents or levels of protein in excess of those already present in wood. Brown rot fungi degrade the carbohydrate fraction of the wood whereas white rot fingi, producing both cellulases and ligninases attack carbohydrate and lignin within wood cells (Liese, 1970; Scheffer, 1973).

Bacterial decay, restricted to wood of high moisture content, is a prolonged process and would not appear to be as important as decay through fungal attack. The role played by the actinomycete grouping of bacteria may be more important than was previously thought (King *et al,* 1978; Baecker and King, 1980). Baecker *et al* (1981), for instance,

found in pure culture studies that decay rates of other organisms could be enhanced or suppressed by the presence of actinomycetes.

A number of extrinsic factors serve to modify the suitability of the wood substrate as an environment capable of supporting the presence of different classes of decay micro-organisms, and the duration and severity of any physical damage caused by them. These factors include nutrient and oxygen availability, moisture content, pH, temperature and competition from other organisms, as well as wood type, an intrinsic determinant of decay susceptibility. All of these factors can be negated by the addition of toxic materials such as wood preservatives.

1.4. The preservation of wood with reference to electricity distribution poles.

The wood substrate must be moist or in conditions of high humidity for fungal decay to occur. Maintaining the moisture content of wood below 20% renders it virtually immune to microbial degradation (Scheffer, 1973). Therefore wood structures such as distribution poles, in constant contact with the predominantly moist soil conditions found in the United Kingdom, are at great risk of microbial degradation and consequent failure, and to extend their viable service life, these structures receive preservative pre-treatment.

Wood preservatives are classed into three groups: preservative oils (e.g. creosote); waterborne preservatives (e.g. copper chrome arsenate); and organic solvent types containing chemicals such as copper and zinc naphthenates (Findlay, 1985). The pre-treatment of electricity distribution poles in the United Kingdom is carried out primarily by the application of creosote oils. Creosote oils are distilled from tar produced during the carbonisation of bituminous coal. The portion of tar boiling from

200°C to 400°C forms the creosote oils used in timber preservation. The oil consists mainly of hydrocarbons, tar acids and tar bases (Wilkinson, 1973).

Before preservative treatment, poles must be seasoned by air drying to a wood moisture content below 24-30%, the 'fibre saturation point'. At fibre saturation point all 'free water' within the cell spaces, the lumina, has evaporated, leaving the wood cell walls still saturated with water. This water is bound chemically and physically to cellulose, hemicelluloses and lignin and its removal during seasoning is slower than that of free water. Well seasoned timber accepts preservatives more freely, while preservative treatment of poorly seasoned timber is inhibited by water already in the cell lumina and by back pressure of trapped air.

Hence, creosote is applied to the pole by one of three vacuum pressure processes: the Rueping empty cell process, the Lowry pressure impregnation process or the Bethell full cell process. The Rueping and Lowry processes coat the wood cell walls of the treated timber with creosote leaving the interior of the wood cells almost empty of the preservative. The Bethell full cell process fills the wood cells with creosote resulting in a loading approximately three times that of the Rueping process.

The Rueping process is the preferred method of pole impregnation used by electrical authorities in the United Kingdom as it requires less preservative and causes less 'bleeding' of creosote to the surrounding soil in warm weather conditions. The process consists of several stages. Initially, compressed air is injected into a sealed treating vessel, containing seasoned poles, forcing air into the wood cells. The vessel is then flooded with creosote at a temperature of between 65°C and 100° C. The air pressure is raised forcing preservative into the wood and further compressing the trapped air. Pressure is released and the greater fraction of preservative is forced out of the wood cells by expanding air. Surplus preservative is drained off and stored for later use. A

final vacuum is applied ensuring that the timber is free of dripping creosote when removed from the treatment vessel (Wilkinson, 1979).

Pressure creosoting can provide protection against decay for any length of time that is likely to be required by any of the normal engineering processes (Smith and Cockcroft, 1967 a), a period according to Chambers (1963) of thirty to fifty years. However, creosoted poles can fail and require replacement much sooner due to internal and external decay by basidiomycetes and soft rot fungi respectively.

1.5. Internal and external decay of creosoted distribution poles.

Decay in distribution poles predominates at the groundline (Chambers, 1963; Smith and Cockcroft, 1967 b, c; Anon, 1971; Becker, 1976) where moisture and oxygen conditions combine to provide an environment conducive to the growth of decay fungi and, in the case of external soft rot, a ready source of inocula present in the soil.

External decay of the pole surface is due to a reduction in the protective effectiveness of creosote treatment over time in service. The speed with which this reduction takes place is considered to be dependant on the initial quantity of creosote applied (Smith and Cockcroft, 1967 a), the quality of creosote applied and 'ageing' of creosote itself (Dickinson *et al,* 1992). Dickinson *et al* (1992) in a study of 526 poles found 19 % had visible signs of soft rot in the wood cell walls of the outer 1 cm of pole material. Smith and Cockcroft (1967 a) argued that a large percentage of distribution poles in the United would fail, due to external decay, before their intended service life was fulfilled.

Internal decay of creosoted poles occurs when the creosote treatment fails to penetrate completely the susceptible sapwood, due to inadequate treatment at the plant or because the wood was not properly seasoned beforehand (Smith and Cockcroft, 1967 a). If internal pole sapwood is left untreated there is a high risk of it becoming infected with spores of wood rotting fungi which gain entry to the poles via checks and splits caused by expansion and contraction stresses in the wood during drying in service. In the United Kingdom, internal decay of creosoted distribution poles is caused primarily by the basidiomycete ***Neolentinus lepideus*** (*Fr*) (Cartwright and Findlay, 1958; Bruce, 1983). This is attributed to its being rather more tolerant of creosote than other wood rotting basidiomycetes (Smith and Cockcroft, 1967 c).

Untreated sapwood has virtually no natural resistance to attack by wood destroying fungi and, when infection occurs, internal decay will develop whenever the wood becomes moist enough. When decay has become established in untreated sapwood, it is not uncommon for it to spread into the more resistant heartwood (Smith and Cockcroft, 1967 b). In an examination of 220,000 creosoted pine poles in Germany, it was established that inadequate creosote penetration always resulted in internal decay (Warrelmann, 1956), and Smith (1955) reported that it had been necessary to replace a large number of creosoted transmission poles after only 10-15 years due to internal decay of untreated sapwood.

External decay would appear to be potentially the more serious decay problem. The outer 2-3 inches of shell in a typical distribution pole contains 80-90% of its bending strength (Bingel, 1988). A quarter inch decay of the outer shell of a 10 inch diameter pole reduces its strength by 14% but a central core of 6 inches in diameter would have to be removed to have the same effect (Smith and Cockcroft, 1967 b). Extensive internal decay can therefore occur before the strength of a pole is materially effected (Smith and Cockcroft, 1967 b).

However, external decay is not perceived to be as great a threat to existing distribution pole stocks as internal decay, for a number of reasons. External decay occurs only after fairly long service life (Smith and Cockcroft, 1967 a). Dickinson *et al* (1992), for instance, referred to a survey of poles carried out by Wylde (1987) which indicated that poles over 36 years old were at greatest risk of soft rot external decay. Remedial treatment of external decay is technically simple and can be very effective because the decaying zone is easily seen, readily accessible and localised in the vicinity of the groundline. Smith and Cockcroft (1967 a) described one method in which a creosote emulsion was applied to the pole surface and sealed with a plastic lined bandage. This treatment successfully increased the creosote loading in the outer zone of the pole to delay the onset or continuation of external decay. However, internal decay, though again primarily found at the groundline, can occur anywhere in the pole where untreated sapwood is present. It can occur early in a pole's service life with no visible signs and the shorter the viable service life of a distribution pole, the greater is its economic loss. Remedial groundline treatments have therefore commonly been employed to control internal decay.

1.6. Remedial treatments for control of internal decay of creosoted distribution poles.

1.6.1. Borates, fumigants, physical supports and biological control.

The remedial application of highly concentrated borates in either solid or liquid form has been under development for treatment of utility poles (Friis-Hansen, 1987; Dickinson *et al,* 1988). However, the treatment is dependant upon high wood moisture contents. According to Becker (1976) the minimum wood moisture content required for most diffusive processes is the fibre saturation point, i.e. about 30 % moisture content

and, for practical purposes, citing Amemiya (1955), should be 50 % moisture content.

Bruce (1983) cited several methods for remedial treatment of poles involving injection of volatile fumigants such as methyl bromide, chloropicrin and sodium N-methyldithiocarbamate dihydrate, and its active breakdown product, methyl isothiocyanate. Bingel (1988) has argued that when fumigants are applied to existing decay voids within the pole, the toxic vapours can dissipate through them and associated checks and cracks, thereby losing their effectiveness. This, together with the highly toxic non specific nature of these products, is probably responsible for their lack of use in the United Kingdom.

High strength low alloy steel physical supports designed to restore the bending strength of wood poles decayed at the groundline are successfully used in the United States (Bingel, 1988) though are not in widespread use in the United Kingdom. The use of such supports necessitates groundline application of chemical preservatives to eradicate incipient decay prior to installation. Smith (1989) recognised this problem with the use of such a system in the United Kingdom.

Biological control methods e.g. in-situ 'application' of microbial antagonists to basidiomycetes, such as ***Neolentinus lepideus*** (*Fr*), to control internal decay in creosoted distribution poles as an alternative to chemical preservatives has received much study with favourable results. ***Trichoderma*** spp. are presently the most utilised types of fungi in the field of biological control. The main problem associated with their use is in establishing and maintaining populations within the pole sufficient to give long term protection. A useful review of developments in this field is given by Bruce (1992).

1.6.2. Groundline application of waterborne fluoride preservatives.

The electricity supply industry has for many years attempted to control internal decay at the groundline of distribution poles by the use of waterborne fluoride preservatives as a remedial treatment (Steinherz, 1939; Becker, 1973, 1976). Fluorides and especially hydrogen fluorides are highly favoured as the wood moisture content required for effective diffusion of these chemicals is considerably lower than for other preservatives such as boron (section 1.6.1.). This is due (Becker, 1976) to the gaseous diffusion of hydrogen fluoride which allows deep penetration of the fluoride ions.

Treatments using toxic pastes of sodium fluoride and bifluorides applied to the pole surface in the groundline region and covered in roofing felt or plastic films sealed with bitumen were superseded by bandages (Chambers, 1963; Becker, 1976) containing a salts mixture of sodium fluoride, bifluorides, dichromates, dinitrophenol and arsenic compounds sealed with a polythene cover. The addition of chromium was originally to reduce corrosion induced by fluoride compounds (Becker, 1973; Wilkinson, 1979). However, after the chromium salt was shown to 'fix' copper in treated wood, its content in the formulation was increased from 5 to 35 % in order to make treated wood more resistant to leaching (Wilkinson, 1979). Arsenic compounds were added for protection against wood destroying insects, especially termites, and were also found to improve resistance to leaching (Becker, 1973). Toxic mixtures of fluoride, known as FCAP (Fluoride / Chromate / Arsenic / Phenol) preservatives (Wilkinson, 1979), are used for remedial treatment of poles in Europe (Becker, 1976; Graf and Zgraggen, 1988) and the United States (Preston, 1988).

The 'Cobra' process for groundline injection of these preservative mixtures originated in Europe in 1937 and has been used there since that date (Smith, 1989). This process was introduced to the United Kingdom in 1947 and operated commercially up to

1988, during which time in excess of 1.5 million in service poles were treated (Smith, 1989).

Application is carried out by forcing preservative paste into the pole to a depth of 6.5 cm via a hollow injection needle propelled by a mechanical pump. Pole diameter determines the number of injections, up to 120, applied in a defined pattern to the pole in a treatment zone which extends approximately 35 cm above and below the groundline. A bitumen coating is applied as waterproofing to the entire treated area and an aluminium sheath is fixed around the treated region above the groundline (Sinclair *et al,* 1991). According to Perrin (1978), cited by Goodell and Pendlebury (1990), the strength loss associated with incising in utility poles is not significant.

The 'Cobra' process introduces a preservative salt paste through the creosoted band into the susceptible uncreosoted sapwood and adjacent heartwood. By the diffusion of the toxic fluoride component of these preservatives throughout the groundline region of poles, incipient decay is prevented or checked. The use of the original salts formulation for remedial injection of poles, DFA or 'Cobra' salts containing dinitrophenol, sodium fluoride and arsenic (III) oxide, was discontinued in 1986 in the United Kingdom owing to concerns over health and safety regarding the arsenic and dinitropehnol components of the preservative.

A new formulation of the waterborne preservative Rentex was adopted for use; again its fungicidal component is fluoride, present as sodium fluoride (11.3 % m/m) and ammonium bifluoride (21.0 % m/m). Sodium dichromate (27.8 % m/m) is included as a 'fixative', though its mode of action is not well understood but may be by the formation of insoluble fluorine-chromium complexes in treated wood. Sodium sulphate (7.5 % m/m) and sodium carbonate (7.5 % m/m) are included as a drying agent and pH stabiliser (pH ~ 6.5) respectively.

Though effective diffusion of fluorides has been well documented for preservative mixtures containing fluorides and bifluorides (Chambers, 1963; Smith and Cockcroft, 1967 b; Becker, 1973; Henningson and Nilsson, 1975; Nijman, 1989; Goodell and Pendlebury, 1990; Sinclair *et al,* 1991), so has their tendency to leach (Smith and Cockcroft, 1967 b, c; Henningson and Nilsson, 1975; Becker, 1976; Sinclair *et al,* 1991, 1993; Smith *et al,* 1993), which varies widely depending upon the preservative formulation, application method and exposure of the treated wood.

1.7. Aims and layout of the present study.

No information exists regarding the diffusion, permanence or environmental impact for this formulation of Rentex in its recent use as a remedial treatment for the groundline injection of creosoted distribution poles. This lack of information concerning Rentex was, until recently, symptomatic of a general tendency within the wood preservative industry to ignore the need for such a breadth of information even with regard to well established wood preservative treatments. However, while the European Standardization for Wood Preservation Progress Report 91-92 (Hue, 1992) indicates recent advances towards the development of new efficacy tests more representative of treatment methods and end-uses of preservatives, progress towards standardised methods for environmental assessment of preservatives is not so well defined, possibly due to the paucity of information from past studies.

In the light of these circumstances the object of this study, to examine both the efficacy and environmental impact of Rentex remedial treatment, represents a departure from purely efficacy based studies of wood preservatives and required to be carried out essentially without recourse to a standardised methodology. This challenge presented an opportunity for an empirical approach to much of the experimental work. As such, the

development latterly of a novel methodology principally to examine the environmental aspects of the preservative in use (chapter 3), became integral to the aims which it was developed to satisfy. The initial aims of the work were as follows:

1. To determine toxic concentrations of Rentex required to control decay and non-decay micro-organisms commonly found in distribution poles.

2. To evaluate preservative diffusion from the sites of injection throughout the groundline region of poles, to establish whether toxic concentrations are achieved and maintained.

3. To monitor the establishment and maintenance of a toxic effect to inhabitant micro-oganisms of remedially treated poles in service.

4. To examine the permanence of toxic preservative constituents in remedially treated poles.

5. To identify any environmental contamination associated with treated poles in service.

A series of field and laboratory experiments designed to fulfill these aims are contained within chapter 2, where these experiments are considered purely with respect to the efficacy of the treatment. Preservative permanence will be quantified by the loss of toxic preservative constituents from treated timber (**2** and **4**) and deposition in adjacent soil (**5**). The contamination of soil associated with remedially treated timber (**5**), with respect to the environmental impact of the treatment, is considered in chapter 3.

The aims of this assessment were as follows:

1. To identify the potential hazard presented by leached Rentex constituents towards specific environmental systems within the domain of remedially treated poles.

2. To construct a physical field model designed to:

a) Incorporate those natural systems identified as probable indicators of detrimental environmental effects associated with the use of Rentex treated timber.

b) Facilitate laboratory study of these natural indicators under conditions comparable with severe field exposure to remedially treated service poles.

c) Evaluate the specific environmental hazard presented by Rentex remedially treated timber.

3. To evaluate the model system for further development toward its useful addition to wood preservative testing protocols.

This programme of research was designed to identify harmful environmental effects of a toxic substance in its specific end-use under simulated field conditions. The ultimate aim of such research must be to obviate traditional long term field assessments of hazardous substances generally, by providing a system incorporating or able to accomodate elements necessary for a wide variety of applications.

CHAPTER 2.

FIELD AND LABORATORY EXPERIMENTATION TO DETERMINE THE EFFICACY OF RENTEX REMEDIAL TREATMENT OF CREOSOTED DISTRIBUTION POLES.

2.1. INTRODUCTION.

2.1.1. Efficacy of the remedial wood preservative Rentex.

The efficacy of leachable boron/fluoride wood preservatives is based on two properties, namely toxicity and permanence. The active preservative component or components must be present at concentrations or loadings sufficient to confer toxicity against the decay organisms common to the wooden structure concerned. Toxic concentrations, which are generally given in kg of preservative per m^3 of wood or as % w/w, must also remain in the substrate such that long term protection is established.

For the remedial use of this Rentex preservative formulation (sections 1.6.2. and 2.2.1.2.1.) in distribution poles, efficacy is dependent upon distribution of the fungicidal component fluoride, present as sodium fluoride and ammonium bifluoride, throughout the susceptible groundline region. Concentrations of fluoride must ensure that the growth of wood destroying basidiomycetes is eliminated. Subsequent maintenance of a toxic effect will depend upon the extent to which the long term stability of toxic fluoride concentrations, to effects such as leaching by weathering, is conferred by the fixative action of the sodium dichromate component.

2.1.2. Rentex preservative application: the injection process.

Application of Rentex paste to a zone 35 cm above and below the groundline of poles is carried out using a series of staggered lines of injections around the pole circumference (Plate 2.1). The distance between each vertical puncture is approximately 10 cm with each vertical line of injections separated by about 5 cm. This provides a thorough application of preservative to each pole in a diamond pattern of injections without significant mechanical

damage.

As the treatment is intended to prevent decay of uncreosoted sapwood and heartwood, the migration of fluoride from the injection sites throughout this region of the pole is central to its efficacy. The internal surface of each injection site provides the surface of the wood through which penetration must occur. Initial penetration of the wood around each injection site will be effectively a pressure impregnation process, the efficiency of which will depend on the physical properties of the wood of each pole allied to the injection process itself.

The injection pump comprises a preservative reservoir in the form of a hollow metal lever arm attached to an injection head consisting of a vertically flattened injection needle below a spring loaded piston (Plate 2.2). A fulcrum is provided by a chain round the pole attached to each end of a metal yoke above the piston. As the lever arm is pulled downward, the needle is forced into the pole and, simultaneously, the preservative paste is forced through the needle. The cumulative pressure applied to the injected preservative is a combination of this injection pressure and the force applied to the preservative by the elastic recovery of the deformed wood of each injection site as the needle is extracted. As the needle is extracted a portion of the injected preservative is typically ejected to the surface of the pole (Plate 2.1) and it is likely that the larger fraction of the pressure applied is expended in this way. The remainder will act to force preservative into the wood surrounding the injection site. Whereas that portion of the pressure applied to the preservative by the injection mechanism will remain relatively constant between each injection site and each pole treated, the force applied by the elastic recovery of the injected timber may vary considerably depending on the groundline moisture content of each pole.

Above fibre saturation point, wood moisture content has no effect on the elastic properties of wood, but decreases in moisture content below this level result in increased elasticity (Kollman and Cote, 1968; Dinwoodie, 1989). Gerhards (1982), cited by Dinwoodie (1989), found that the modulus of elasticity increased by 1.5 % per unit

Plate 2.1. Injection of Rentex preservative paste into the groundline region of a creosoted pole section using an injection pump.

Plate 2.2. The head of the injection pump showing the injection needle and piston below a metal yoke for chain attachment to the pole.

decrease in moisture content within the range of 6 - 20 % wood moisture content.

Wood with a higher moisture content and therefore lower elasticity will impose less pressure for penetration on injected preservative than wood of a moisture content below fibre saturation point and consequently greater elasticity. In addition, due to the presence of free water in the tracheids and cell lumen, the preferential path of liquid flow (Wardrop and Davies, 1961), any pressure applied by wood above the fibre saturation point may result in only a small degree of preservative penetration. However, wood of moisture content below fibre saturation point will be provided with ample voids into which preservative may be forced by the greater pressure applied in this situation. Therefore, the lower the wood moisture content below fibre saturation point the greater the pressure for penetration. Conversely, the greater the wood moisture content above fibre saturation point the greater the resistance to penetration.

Groundline moisture content of poles in service may therefore have a major influence on initial uptake of preservative to the surrounding wood at each injection site and also on the total loading of preservative in each pole.

2.1.3. Migration of fluoride in Rentex treated timber.

After initial movement of preservative into wood surrounding each injection site, further penetration of preservative components will take place and this process will also be greatly influenced by wood moisture content. Whereas initial penetration of wood by injection of a given quantity of the applied preservative mass is due to a pressure gradient resisted by the presence of free water in the cell lumen, diffusion is the movement of molecules or ions of the preservative in response to concentration gradients and is enhanced by the presence of free water. Diffusion in the interior of the wood should continue until equalization of concentrations at some depth is achieved (Kollman and Cote, 1968; Nijman, 1989).

The main precondition for the diffusion of most preservatives applied by non-pressure methods is a wood moisture content of at least fibre saturation point, approximately 30 % (Becker, 1976) or, according to Amemiya (1955), cited by Becker (1976), a minimum of 50 % under practical conditions. For this reason, the diffusion process was originally applied to freshly felled green or unseasoned wood, though later it was realised that after rewetting, seasoned timber could also be treated satisfactorily by the same diffusion process (Becker, 1976).

The groundline moisture content of individual distribution poles within a population from any given geographical area will differ due to topographical position, surrounding soil drainage, degree and location of pole surface checking, efficiency of creosote treatment and erection date. Seasonal rainfall and temperature effects will result in smaller intra-pole variations in moisture content throughout the year. Consequently, there was wide variation in the moisture contents of service poles used in the present study and a substantial proportion of these were found to be below the fibre saturation point. However, the diffusion of fluoride preservatives can occur at moisture contents very much lower than fibre saturation point due to the gaseous diffusion of hydrogen fluoride. Gaseous diffusion is more pronounced for hydrogen fluoride salts such as ammonium bifluoride than for sodium fluoride or fluorosilicates, and is encouraged by acidic conditions (Becker, 1973, 1976; Becker and Berghoff, 1963). Rentex is slightly acidic with a pH of around 6.5 and hydrogen fluoride in the ammonium bifluoride component accounts for one third of the total fluoride content of the preservative. Consequently, at lower moisture contents below fibre saturation point, the diffusion of fluorides in distribution poles may rely heavily on gaseous diffusion, whereas at higher moisture contents above fibre saturation point, the presence of free water will reduce the importance of gaseous diffusion and enhance diffusion in the liquid phase.

The importance of gaseous diffusion and moisture content is indicated by the findings of Becker (1959), cited by Becker (1976), when comparing preservative salt concentrations of bifluorides and magnesium fluorosilicate in pine stored for five years after non-pressure

application. In wood which was air-dried to 9 % moisture content prior to treatment, penetration depth of magnesium fluorosilicate, at 10 mm, was half that of the bifluoride mixture and half that achieved by the same fluorosilicate applied to wood at 50 % moisture content. A similar pattern of higher penetration for the bifluoride mixture was displayed at the higher moisture content. These findings confirm those of Buro and Becker (1956) where diffusion of sodium fluoride was shown to occur at wood moisture levels of as little as 17 % and below and the rate of diffusion displayed a linear increase with increasing wood moisture content. Further confirmation is provided by Liese and Schubert (1941) cited by Kollman and Cote (1968) where a greater longitudinal diffusion rate of sodium fluoride was apparent in green unseasoned pine sapwood compared to air dried sapwood.

That the greater penetration of bifluoride found by Becker (1959) compared to magnesium fluorosilicate was due largely to enhanced gaseous diffusion of hydrogen fluoride is supported by the similar solubilities of both fluoride compounds (Becker, 1973), which would indicate that at a given wood moisture content penetration should be similar. Though both preservatives at each moisture content displayed a reduction in concentration as penetration increased, this was less pronounced for the bifluoride, which resulted in a more even distribution of fluoride through the wood. A more even distribution of both preservatives was found for the wood of higher moisture content due to the enhanced diffusion conditions.

In addition to wood moisture content; type of fluoride preservative; and increased penetration depths after longer diffusion times (Liese and Schubert, 1941; Buro and Becker, 1956); grain orientation is also an important factor affecting penetration depth. The order of decreasing rate of diffusion according to direction of penetration in pine is longitudinal > radial > tangential for diffusion of sodium fluoride (Liese and Schubert, 1941; Becker, 1973, 1976; Buro and Becker, 1956) and for the diffusion process generally (Christensen, 1951 a, b). Becker (1976) explained these differences in terms of the migration pathways of salts in cell walls, the area of cell wall cross sections through which

the salt may diffuse in the longitudinal direction being approximately twice that of the radial and tangential directions. The greater degree of radial as opposed to tangential diffusion was explained by the presence of the permeable ray cells in this orientation (Becker, 1976; Christensen, 1951 a). Christensen (1951 a) suggested that the relatively higher longitudinal diffusion rate was due to passage through the wider cell lumen in the longitudinal direction as a result of the vertical orientation of the tracheids. This is logical given that Cady and Williams (1934) were able to characterise diffusion in saturated pine blocks according to the mechanical restraint imposed by the radii of pores in tracheid pit membranes, which would hinder longitudinal diffusion to a lesser extent due to their less frequent occurrence in this diffusion pathway.

Penetration depths achieved by the fluoride components of Rentex in the remedial treatment of service poles is not known. However, some indication of these penetration depths may be indicated by findings from a laboratory test of a Rentex formulation for interior building use. Immersion of spruce planks at 12 - 14 % moisture content in 12 % preservative solutions for periods of 2, 4 and 16 hours initially gave penetration into the end grain of 10.8, 12.1 and 15.8 mm respectively for fluoride (Anon, 1966). This initial penetration was not a diffusive process but rather was caused by hydrostatic pressure which is a function of the depth of immersion and capillary pressure, itself a function of preservative solution surface tension, the contact angle and the diameter of capillaries in the wood (Smith and Purslow, 1960). After initial penetration, the planks were wrapped in plastic for 3 weeks to allow diffusion to occur, and end grain penetration, for planks subjected to original immersions of 2, 4, and 16 hours, had risen to 17.8, 19.8 and 27.0 mm respectively. Initial penetration diagonal to the wood fibres was as expected lower, giving 2.7 and 4.9 mm for planks immersed for 2 and 16 hours respectively, rising to 5.4 and 7.5 mm after 3 weeks diffusion. Although these findings relate only to spruce they are nevertheless encouraging for remedial treatment of the predominantly Scots pine pole stocks of the United Kingdom, since Buro and Becker (1956) have shown more efficient diffusion of fluoride compounds in pine compared to spruce.

Field studies of groundline injected DFA salts, containing dinitrophenol, sodium fluoride and arsenic (III) oxide, used in the United Kingdom prior to the adoption of Rentex, indicated good fluoride distribution throughout the groundline region of poles (Panek *et al,* 1961 cited by Smith and Cockcroft, 1967 c; Smith and Cockcroft, 1967 b, c). Similar findings were obtained by Graf and Zgraggen (1988) for injection of a 'Cobra' salt formulation containing sodium fluoride, sodium dichromate and dinitrophenol, and by Goodell and Pendlebury (1990) in a study of ammonium bifluoride and sodium fluoride diffusion from centre bored sites through the butt ends of wood poles.

Clearly then, there is ample evidence indicating that substantial diffusion of fluorides from the injection sites in Rentex treated field poles is likely to occur. However, no studies have been carried out using this particular preservative formulation and treatment method to confirm these indications. Such a study must therefore form a part of any experimental programme intended to examine the efficacy of Rentex treatment.

2.1.4. Effective toxic concentrations of fluoride in Rentex treated timber.

Migration of fluoride from injected Rentex preservative must ensure that concentrations toxic to the decay organisms are distributed throughout the susceptible groundline area of treated poles. The basidiomycete most commonly associated with internal decay of creosoted distribution poles in the United Kingdom is ***Neolentinus lepideus*** (Fr.) (Cartwright and Findlay, 1958; Bruce, 1983; Smith and Cockcroft, 1967 c). Therefore the concentration of fluoride necessary to control this organism, could reasonably be taken as the minimum required to indicate the efficacy of diffused fluoride concentrations of Rentex in wood. This is in keeping with the traditional approach for determining effective loadings or concentrations of preservatives.

Standard laboratory toxicity tests carried out for earlier formulations of Rentex for interior building use (Bunker and Findlay, 1964; Anon, 1963, 1964 a, b) and original 'Cobra' pastes (Findlay, 1947; Anon, 1986 a, b) are of little use as a guide to the effective toxicity of Rentex for remedial treatment because no indication was given of the precise formulation of preservative used. In consequence the toxic concentration of fluoride cannot be calculated. However, these tests when taken as a whole indicate a relative lack of tolerance of ***Neolentinus lepideus*** to fluoride concentrations, which was previously shown by Richards (1924) in a comparative study of 17 species of wood destroying fungi.

Becker (1973), drawing on a variety of sources, indicated that concentrations of fluorides or bifluorides of 1 kg/m^3 of wood, was sufficient to ensure protection of wood against fluoride tolerant wood destroying basidiomycetes. A similar figure for fluoride was quoted by Henningson and Nilsson (1975) from the work of Liese and Groger (1954). However, the toxic values of a wood preservative are ordinarily expressed as 2 concentrations; the upper toxic limit corresponding to the lowest preservative concentration preventing decay, and the lower toxic limit corresponding to the highest preservative concentration at which the wood is no longer adequately protected (EN 113: BSI 1982). Smith and Cockcroft (1967 c) for instance, gave a toxic range for fluoride against ***Neolentinus lepideus*** of 0.08 - 0.32 kg/m^3 based on the findings of Van der Berge (1934), Liese et al (1935) and Savory (1956).

Smith and Cockcroft (1967 b, c) examined the diffusion of fluoride in creosoted distribution poles at 2, 4, 10 and 11 years after remedial groundline treatment with 'Cobra' salts containing sodium fluoride, dinitrophenol and arsenic (III) oxide, using this toxic range to determine whether the distribution of fluoride in the poles was sufficient to prevent internal decay.

At 2 years after treatment, 4 wood cores spaced 90° apart round the circumference of each of 5 poles were recovered from 6 inches above and below the groundline avoiding

'Cobra' incisions. Each core was cut into five 0.5 inch lengths measured from the pole surface. The corresponding core lengths from each level of each pole were combined and chemically analysed in duplicate for fluoride content using the colorimetric method of Megrigian (1954). Mean fluoride contents were generally well above the accepted toxic range of 0.08 - 0.32 kg/m^3 of wood, the lowest concentration being 0.32 kg/m^3 found above the groundline at a depth of 2.5 inches. Combined means for all poles and all core lengths above and below the groundline were 1.12 and 1.35 kg/m^3 respectively.

Eight equidistant cores were removed from 6 inches above and below the groundline of each of 5 poles which had been remedially treated 10 years previously. Each 0.5 inch of core length was analysed separately for fluoride content using the more sensitive colorimetric method of Greenhalgh and Riley (1961) to take account of smaller sample size and the lower levels of fluoride likely to be present. Half of the 8 cores at each height were analysed in this way to give radial fluoride loadings. Approximately half of the samples had fluoride contents at or below the accepted toxic level representing a substantial decrease in comparison to the 2 year period. Variation in fluoride content tended to decrease as core sample depth increased. Therefore the tendency for fluoride contents of zones which were below the toxic range to be masked by others to give mean loadings at or above this range decreased as sample depth increased. Consequently, in order to improve evaluation of the decrease in fluoride concentrations in poles over a given period, a single core from each of 2 groups of 20 poles treated 4 and 11 years previously was removed and that portion at 2 - 2.5 inches from the pole surface was analysed. The combined mean fluoride content of samples from poles, 11 years after treatment, was 0.14 kg/m^3; however, this figure included only 3 values above the toxic range whereas 13 were below it. The more representative median value of 0.016 kg/m^3 was substantially below the toxic range. The mean fluoride content and standard deviation of samples taken 4 years after treatment was 0.85 kg/m^3 and 1.02 kg/m^3, indicating that even at this stage an appreciable number of poles were likely to contain zones at or below the accepted toxic range. Based on these findings, Smith and Cockcroft (1967c) expected 'Cobra' treatment to give protection against internal decay in

the groundline zone of poles for approximately 8 - 9 years, though it was stressed that the period of protection would vary widely between individual poles.

The greater penetration of bifluorides (Goodell and Pendlebury, 1990; Becker, 1959, 1973, 1976) and more even distribution in wood (Buro and Becker, 1956), section 2.1.3., would indicate that their presence in Rentex in addition to sodium fluoride, the exclusive form of fluoride in 'Cobra' salts, may result in an improvement in fluoride distribution leading to an enhanced period of protection. However, to judge the efficacy of a preservative field treatment solely on laboratory indications of the preservative's toxicity to a target organism is unwise and should be supported by studies of efficacy against organisms in field poles.

Bruce and King (1989) carried out such a field study using a formulation of Rentex containing ammonium bifluoride and sodium fluoride. Creosoted pole sections were artificially inoculated with ***Neolentinus lepideus*** prior to 'Cobra' injection. Fifteen months after treatment no re-isolations of the decay fungus were made from these poles, whereas in the majority of inoculated control poles which had received no Rentex treatment, re-isolations were common. Though these findings are valid, their specific relevance to the efficacy of the Rentex remedial treatment under study is in doubt due to the unspecified formulation of Rentex used and indications, on pole sectioning (section 2.2.6.4.), that an excessive number of preservative injections were made to these pole sections (section 2.3.6.2.).

2.1.5. Permanence of fluoride concentrations in Rentex treated timber.

For any fluoride based preservative reliant on the diffusion process, the concentration of diffused fluoride necessary to protect the treated timber from decay must be maintained over an extended period in order to make the treatment economically viable. For fluoride

preservatives of timber for use in ground contact outdoors, impermanence of preservative at concentrations above toxic levels is primarily due to leaching (Becker, 1973, 1976; Smith and Cockcroft, 1967 b, c; Henningson and Nilsson, 1975; Sinclair *et al,* 1993; Smith *et al,* 1993).

The addition of chromates and dichromates to waterborne fluoride preservatives, originally intended to reduce iron corrosion induced by fluoride compounds, was found to inhibit leaching of fluoride from treated wood (Becker, 1973, 1976; Wilkinson, 1979). Chromium is present in Rentex as sodium dichromate, the highly soluble chromium (VI) form. This form of chromium is present in copper chrome arsenate (CCA) waterborne preservatives which are applied by pressure impregnation and for which 'fixation' and resistance to leaching are well known (Wilkinson, 1979; Nicholas, 1972; Kumar and Morell, 1988; Cooper, 1988; Yamamoto and Ruddick, 1992), though the precise mechanism of fixation is not fully understood.

During the reaction with wood, chromium (VI) is 'fixed' (Feist and Ellis, 1978) and reduced to the more stable insoluble chromium (III) form (Wright and Banks, 1989; Yamamoto and Ruddick, 1992). Chromium (III) complexes with polysaccharide (Nicholas, 1972) and lignin components of the cell wall (Wright and Banks, 1989) and may also react with arsenic to form insoluble chromium (III) arsenate (Nicholas, 1972). In electron spin resonance studies of CCA in treated wood, Yamamoto and Ruddick (1992) found that chromium (VI) was initially reduced to chromium (V) which remained, in addition to chromium (III), in treated wood over a 6 month fixation period. Based on these findings and the extended chromium fixation times determined by McMahon *et al* (1942), Yamamoto and Ruddick (1992) postulated that the reaction of CCA in wood proceeds to a relatively rapid insolubilization reaction driven by chromium oxidation-reduction reactions with possible copper-chrome, copper-arsenate or chrome arsenate complex formation. This is followed by a slow rearrangement or chemical reformation of hydrolysis products which have long term stability and it is these complexes which provide CCA with its performance.

The insoluble fixed chromium (III) form is by far the most stable oxidation state for this element which forms stable salts with all the common anions including fluoride (Greenwood and Earnshaw, 1989). The formation of possibly insoluble fluo-chrome complexes may occur in wood after application of waterborne chromate fluoride preservatives such as Rentex. For instance, a determination of the leach resistance of fluorine and chromium components in wood, vacuum impregnated with 'Fluralsil-Ull', according to the German standard method DIN 52 176, indicated that 35.9 % and 20.5 % of applied fluoride and chromium respectively was leached (Anon, 1960). The formulation of 'Fluralsil-Ull' was stated to be identical to a formulation of Rentex and treated wood was gradually dried for 4 weeks prior to leaching to allow fixation to occur. These findings indicate a degree of resistance to leaching when chromate-fluoride preservatives are, like CCA, pressure impregnated into wood. This application method ensures that the components of a preservative are generally evenly spread and are in intimate contact within the treated wood. However, for the remedial use of Rentex in distribution poles, the proximity of fluoride and chromium components throughout the cross section of the treated groundline area will depend on their respective diffusion rates.

Graf and Zgraggen (1976) noted that 1 year after applying 'Cobra' treatment to unseasoned spruce pole sections, using a paste containing sodium fluoride and sodium dichromate, chromium was restricted to an area 5 mm either side of the injection site. Fluoride had diffused to concentrations above 0.2 % throughout the cross section of the poles with a high concentration around the injection site. This indicates that the respective diffusion rates of fluorine and chromium are quite different even in timber where high moisture content will favour diffusion of both, and that the diffusion of chromium is actively resisted by the fixation process. It is evident from the findings of Graf and Zgraggen (1976) that fixation of fluoride could only occur immediately adjacent to the injection site. Clearly, a similar disparity in the distribution of fluoride and chromium in Rentex treated timber would not favour the long term maintenance of toxic fluoride concentrations. Hence, a knowledge of the distribution of both these preservative components in remedially treated

timber is necessary for an accurate determination of the permanence of fungitoxic fluoride concentrations.

2.1.6. Outline of the experimental programme to evaluate the efficacy of Rentex remedial treatment.

Though certain aspects of the efficacy of a large number of fluoride preservatives have been well documented (sections 2.1.3. - 2.1.5.), few, if any, of these formulations have been subjected to studies designed to examine all aspects of efficacy. With regard to the Rentex formulation under study here, no previous field or laboratory trials have been carried out to determine its suitability as a remedial groundline treatment for creosoted distribution poles. Hence, the following studies were undertaken to provide a complete profile of Rentex treatment in terms of all the characteristic determinants of the efficacy of fluoride preservatives generally (sections 2.1.3. - 2.1.5.).

(1) A microbiological laboratory study carried out to determine toxic concentrations of Rentex and its fluoride component in Scots pine wood blocks necessary to control fungi commonly found in distribution poles. This study determined the required protective concentrations of fluoride in sapwood and heartwood for comparison with monitored levels in the follow-up field study (Section 2.2.1.).

(2) Assessment of the distribution and permanence of fluoride and chromium components of the preservative after periods of field exposure by chemical analysis of wood samples recovered from throughout the uncreosoted groundline area of remedially treated creosoted pole sections (Section 2.2.2.).

To complement and support the findings from the microbiological laboratory studies (**1**) and the field based investigation entailing chemical analysis of treated timber (**2**) the following field based studies were carried out:

(**3**) A microbiological study to monitor any toxic preservative effect on populations of inhabitant micro-organisms isolated from uncreosoted wood between injection sites of remedially treated representative poles in service (Section 2.2.3.).

(**4**) Chemical analysis of wood core samples, removed from remedially treated distribution poles 18 months after treatment, for fluoride and chromium content, to determine whether toxic preservative levels were present (Section 2.2.4.).

To evaluate the permanence of the total fluoride and chromium applied to timber during remedial treatment the following field based studies were undertaken:

(**5**) Measurement of concentrations of fluoride and chromium in soils adjacent to Rentex treated creosoted service poles up to 12 months after treatment (Section 2.2.5.).

(**6**) Determination of the fluoride and chromium content of wood sawdust samples recovered from Rentex treated creosoted pole sections subjected to 0, 2 and 4.25 years field exposure (Section 2.2.6.).

(**7**) Determination of the fluoride and chromium content of field soils adjacent to Rentex treated creosoted pole sections at 2 and 4.25 years after remedial treatment (Section 2.2.7.).

2.2. MATERIALS AND METHODS.

2.2.1. The Toxicity of Rentex impregnated wood to selected fungi.

2.2.1.1. Summary.

An assessment of Rentex preservative effectiveness against a number of fungi commonly isolated from distribution poles was carried out using a method adapted from the European Standard laboratory procedure EN 113 (BSI: 1982). Wood blocks, vacuum impregnated with a range of preservative concentrations were exposed to attack by the fungi in pure culture over a period of 16 weeks. For basidiomycetes, weight losses of the blocks were used to establish the toxic values of the preservative (section 2.2.1.2.6.). For moulds, which caused no appreciable weight loss in wood blocks, a scoring system was devised to determine a toxic value based on the degree of mould cover over the blocks (section 2.2.1.2.7.).

2.2.1.2. Methods.

2.2.1.2.1. Preservative solutions.

Earlier toxicity tests using other Rentex formulations (Bunker and Findlay, 1964; Anon, 1963, 1964 a, b) indicated that, for the fungi examined in the present study (section 2.2.1.4.), the concentration range of preservative solutions used need not exceed 2 %. Preparation of a 2 % w/v preservative solution, carried out using Analar quality reagents and grade A glassware, was based on the original formulation of 'Cobra' Rentex for field use, given as:

Chemical name and formula	% m/m
Anhydrous sodium dichromate ($Na_2Cr_2O_7$)	27.8
Ammonium bifluoride ($NH_4F.HF$)	21.0
Sodium fluoride (NaF)	11.3
Anhydrous sodium carbonate (Na_2CO_3)	7.5
Anhydrous sodium sulphate (Na_2SO_4)	7.5
Water (H_2O)	24.6
Gum tragacanth	0.3

Ammonium bifluoride (4.2000 g) was added to 500 cm^3 of distilled water and dissolved over 5-10 minutes. Anhydrous sodium carbonate (1.5000 g) was added slowly and mixed for a further 5 minutes, then anhydrous sodium dichromate (5.5600 g), anhydrous sodium sulphate (1.5000 g) and sodium fluoride (2.2600 g) were added in that order. Mixing was continued for 15 minutes, during which gum tragacanth (0.0600 g) was slowly added. The solution was mixed until all particulate matter was dissolved, poured and rinsed into a 1 dm^3 volumetric flask and made up to volume with distilled water. Sequential dilutions of this 2 % w/v solution were carried out with distilled water to provide Rentex solutions of 1.00, 0.50, 0.25, 0.10 and 0.05 % w/v. Distilled water represented a 0.00 % w/v solution of Rentex.

2.2.1.2.2. Wood blocks.

Dimensions and density.

Quartersawn sections of Scots pine (***Pinus sylvestris*** Linnaeus), obtained from the City of Dundee Parks Department, were dried, in a fan oven at 40°C for three weeks, to constant weight. Wood blocks of 50 mm x 25 mm x 15 mm were cut separately from the sapwood and heartwood zones, avoiding a completely tangential orientation of the growth rings on

the broad faces and with longitudinal faces parallel to the direction of the grain. The blocks were placed overnight in an oven at 105°C, removed, cooled in a glass desiccator and dry weights recorded. Ten percent of the sapwood and heartwood blocks were retained for determination of respective mean densities, by the water displacement technique of Desch (1977). The mean densities of sapwood and heartwood blocks used in the study were 527.5 and 398.6 kg/m^3 with standard deviations of 55 and 49 kg/m^3 respectively.

Preservative impregnation.

For each preservative concentration, wood blocks, were piled cross-wise and ballasted with glass microscope slides in a plastic treatment vessel within a vacuum desiccator attached to a vacuum pump. The pressure within the desiccator was reduced to 7 mbar, held for 15 minutes and a stopcock to the vacuum pump closed. Another stopcock, attached to a reservoir of the preservative solution, was opened, thereby drawing the solution into the treatment vessel and completely submerging the wood blocks. Air was admitted to the vacuum vessel. The treatment vessel was removed and left for 2 hours, during which time further preservative solution was added to keep the wood blocks fully covered. Wood blocks were individually removed from the treatment vessel, excess liquid was removed by light blotting with absorbent paper and each block was immediately weighed to determine the mass after impregnation. The wood blocks were dried at 20-22°C in a glass covered shallow wooden box, their narrow faces resting on a plastic mesh. The wood blocks were inverted every 3 days and, after 2 weeks, the box was progressively uncovered until, at 3 weeks, the glass sheet was discarded and drying continued for 1 more week.

Sterilisation.

Sterilisation of all wood block specimens was carried out by exposure to ethylene oxide vapour. Wood blocks were placed around a 50 cm^3 capacity glass beaker in a glass

desiccator, free of desiccant. A laboratory fume cabinet was washed down with alcohol and the desiccator placed inside. The sterilant (25 cm^3) was added to the beaker and the desiccator and cabinet immediately sealed. Volatilisation of the sterilant proceeded rapidly and the wood blocks were left in contact with the fumes for 1 day. The desiccator was opened, the cabinet resealed and the specimens ventilated for 2 days with filtered air drawn into the cabinet from the laboratory and expelled outside.

Distribution of wood blocks.

Blocks impregnated with preservative solutions formed the *treated test specimens* and the *treated check test specimens.* The former group, 4 blocks for each of the 7 preservative concentrations (section 2.2.1.2.1.) were subjected to attack by each of the 6 fungi (section 2.2.1.2.3.). The latter group was exposed to identical conditions but were not exposed to fungal attack. These blocks, 4 for each preservative concentration, were used for determination of the correction factor for mass changes resulting from factors other than basidiomycete attack. A further group of blocks impregnated with distilled water only, formed the *control specimens* and *virulence control specimens. Control specimens,* 1 placed between 2 *treated test specimens* per culture vessel (section 2.2.1.2.4.), were subjected to fungal attack (section 2.2.1.2.5.) alongside the *treated test specimens. Virulence control specimens,* 6 for each basidiomycete and 4 for each mould, were subjected to fungal attack in order to confirm the virulence of each strain used.

2.2.1.2.3. Fungal cultures.

All fungal cultures were obtained from the collection at the University of Abertay Dundee. The basidiomycetes chosen for the test were the brown rots ***Neolentinus lepideus*** (BAM 20), ***N. lepideus*** (pole isolate 4) and ***Coniophora puteana*** (BAM 15), which were tested against sapwood and heartwood wood blocks. The white rot ***Coriolus versicolor*** (FPRL 28B) was tested against sapwood blocks only. Similarly the moulds, ***Trichoderma***

polysporum (IMI 206039) and ***Cladosporium resinae*** (pole isolate), were tested against sapwood blocks only. Those fungi identified as pole isolates were strains collected from field poles during previous research studies at the university. All fungi were maintained in petri dishes on malt extract agar prepared as for the culture vessels (section 2.2.1.2.4).

2.2.1.2.4. Growth media and culture vessels.

Growth media consisted of malt extract Oxoid L39 (40 g) and Agar A, Oxoid L11 (25 g) dissolved and made up to volume with distilled water in a 1 dm^{-3} volumetric flask. Approximately 200 cm^3 were poured into glass culture jars, of diameter 90 mm and height 85 mm, which were sealed with metal lids pierced with a foam insert to allow aeration, and autoclaved. On cooling to approximately 50°C, Benomyl and Streptomycin sulphate were added aseptically to give final concentrations of 4 ppm and 0.001 g/cm^3 respectively (Clubbe and Levy, 1977), to prevent contaminating growth of moulds and bacteria respectively. No Benomyl was added to media intended for mould growth.

2.2.1.2.5. Introduction of fungi and wood blocks to culture vessels.

On cooling and solidification of the media, 3 cores from each basidiomycete petri dish culture (section 2.2.1.2.3.) were transferred aseptically to the prepared culture vessels. After 3 weeks growth, the wood blocks were inserted, supported on sterile plastic mesh placed over the cultures. For the moulds (section 2.2.1.2.3.), the wood blocks were introduced first and inoculated directly by spraying a spore suspension of each mould. The culture vessels were maintained at 25°C and 70 % relative humidity for 16 weeks.

2.2.1.2.6. Toxic values of the preservative against basidiomycetes.

Definition of preservative toxic values.

The toxic values of a preservative, as defined in EN 113 (BSI: 1982), are expressed as two preservative concentrations: one corresponding to the lowest preservative concentration protecting the wood and the other corresponding to that preservative concentration immediately below in the series of concentrations used, at which the wood is no longer adequately protected. Protection at a given concentration is regarded as adequate if the corrected mean loss in mass of the *treated test specimens* is less than 3 % of their initial dry mass.

Calculation of the loss in mass of *treated test specimens.*

The corrected loss in mass of each *treated test specimen* subjected to attack by basidiomycetes was calculated for each preservative concentration as follows:

$$\text{Corrected mass loss} = \frac{(M_1 + C) - M_2}{(M_1 + C)} \times 100$$

M_1 = Dry weight (g) of *treated test specimen* prior to basidiomycete attack.

M_2 = Dry weight (g) of *treated test specimen* after basidiomycete attack.

C = The mean change in mass of the *treated check test specimens* over 16 weeks.

Calculation of the retention of preservative in wood blocks.

Retention or loading of the preservative product in each treated wood block specimen was calculated in kilograms per cubic metre by the formulae, *1* to *7*,

(*1*) $W_1 - W_2$ = **[A]** Uptake of preservative solution by wood block (g).

W_1 = Weight (g) of wood block immediately after preservative impregnation.

W_2 = Dry weight (g) of unimpregnated wood block.

$$(2)\quad \frac{[A]\ (1)}{\text{Volume of block } (cm^3)} \times \frac{\text{Solution concentration } (\%)}{100} = [B]$$

Where **[B]** = Retention of the product at a given concentration (g/m^3).

$$(3)\quad \frac{\text{Dry weight of block (g)}}{\text{Volume of block } (cm^3)} = [C]$$ Weight of $1cm^3$ block (g).

$$(4)\quad \frac{[B]\ (2)}{[C]\ (3)} = [D]$$ Retention of the product at a given concentration (g/g).

(*5*) **[D]** (4) x 10^6 = **[E]** Retention of product (ug/g).

(*6*) **[E]** (5) x [Mean density of wood (kg/m^3) x 10^3] = **[F]**

Where **[F]** = Retention of the product (ug/m^3).

(*7*) **[F]** (6) x 10^9 = **Retention of product (kg/m^3).**

Retentions of preservative in wood blocks, as concentrations of Rentex and fluoride (% w/w) were calculated for toxic values only as follows:

$$\text{Rentex conc. (\% w/w)} = \frac{\text{[E] Retention of product (ug/g)}}{\text{1,000,000 ug of wood}} \times 100$$

The formulation of Rentex contains 19.1 % fluoride (m/m), therefore

Rentex conc. (% w/w) x 19 / 100 = **Fluoride concentration (% w/w).**

2.2.1.2.7. Toxic limit of the preservative against moulds.

After 16 weeks in contact with the moulds (section 2.2.1.2.5.), the wood blocks were removed from the culture vessels. As mould growth causes no loss in mass in wood blocks, a scoring system was used to measure the degree of mould cover on the blocks: 0, no cover; 1, 25 % mould cover; 2, 50 %; 3, 75 %; and 4, 100 % or total mould cover. That concentration of preservative allowing no mould growth over the *treated test specimens* was taken as the toxic limit of the preservative against mould growth. Retention of the preservative product in treated wood blocks in kg/m^3 and retentions of preservative, as

concentrations of Rentex and fluoride (% w/w) for toxic limits only, were calculated as for wood blocks exposed to basidiomycetes.

2.2.2. Evaluation of the distribution of fluoride and chromium in Rentex treated pole sections.

2.2.2.1. Summary.

Rentex treated pole sections, 6 m long, were erected and maintained at a field site for up to 20 months. Wood samples for preservative distribution studies were obtained from the pole sections (section 2.2.2.2.). An alkali fusion extraction technique, for fluoride analysis of soils and vegetation (McQuaker and Gurney, 1977), was modified (Sinclair *et al*, 1991) for these samples to include extraction of chromium into solution (section 2.2.2.3.3.). Fluoride and chromium contents of the sample solutions were measured by ion-selective electrode and atomic absorption spectrophotometry respectively, using the method of standard additions (sections 2.2.2.3.4. and 2.2.2.3.5.). The modified method was evaluated for accurate determination of the fluoride and chromium levels in wood and soil (Appendix 1).

2.2.2.2. Pole sections.

2.2.2.2.1. Preparation.

Twenty four 6 m pole sections were cut from the mid to upper portions of aged creosoted distribution poles and the circumference of each section recorded. The top end of each was tapered and coated with bitumen to facilitate rainwater run-off. The pole sections were erected to a depth of 1 m in holes excavated in a sandy loam soil at Tealing in the east of Scotland in November 1989 and remedially treated with the preservative by the 'Cobra' process (sections1.6.2. and 2.1.2.). The number of injections applied to each pole section was recorded and the soil excavations refilled.

2.2.2.2.2. Sampling.

After preservative injection (section 2.2.2.2.) 9 pole sections were immediately removed from the site, 7 being set aside for a later study (section 2.2.6.). The treated zones were cut from the 2 remaining sections (figure 2.2.1/1) and these zones were further sectioned at approximately 17.5cm above and below the groundline position (figure 2.2.1/2).

Two contact points on a hand held *Protimeter* conductivity moisture meter, previously calibrated to the % moisture content of wood, were pressed firmly into the uncreosoted centre of each exposed surface. Conductivity and hence % moisture content of the wood was recorded between the 2 contact points.

Two discs approximately 1 cm deep were cut from both surfaces (figure 2.2.1/3) and wood samples were excised from these discs along preservative injection lines at A, B, C and D and labelled 1-8 (figure 2.2.1/4). Wood samples were bulked and finely ground in a hammer mill to provide a representative sample for each position 1-8. This procedure provided wood samples labelled 1-8 from above and below the groundline of each pole section. A further 2 pole sections were recovered and similarly sampled at 2, 5, 12 and 20 months after remedial treatment. All wood samples were analysed for fluoride and chromium content (section 2.2.2.3.).

2.2.2.3. Determination of the fluoride and chromium content of pole section wood samples.

2.2.2.3.1. Apparatus.

Fluoride contents of the sample solutions were measured using a Corning Eel model 12 pH meter equipped with a Russell model 94-4099 fluoride electrode and reference electrode

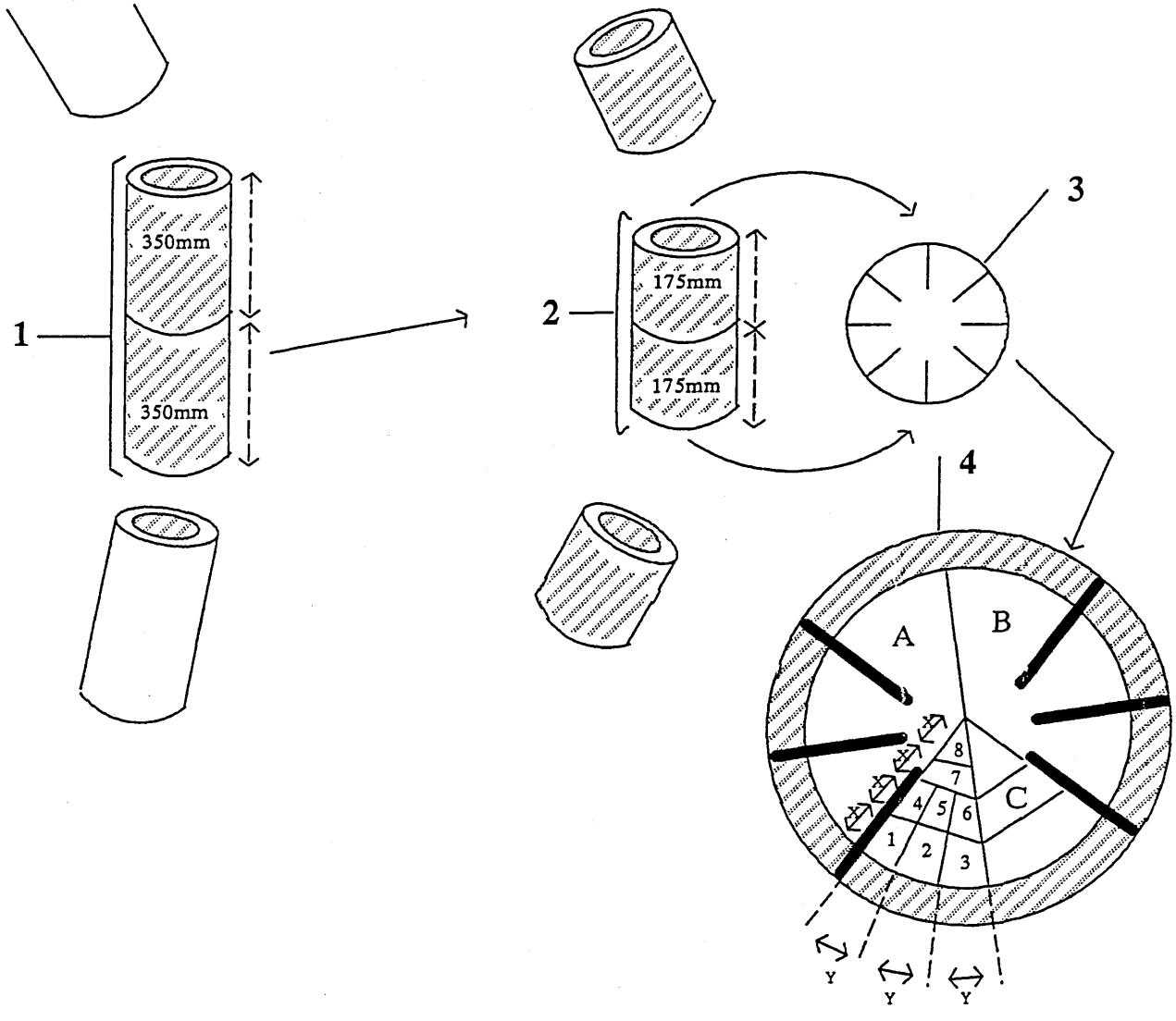


Figure 2.2.1. Procedure for removal of wood samples 1 - 8 from the uncreosoted area of the remedially treated zone of distribution pole sections for fluoride and chromium distribution studies (see section 2.2.2.2.2. for description).

type 900019. Total chromium contents of the sample solutions were measured using a Perkin Elmer 1100B atomic absorption spectrophotometer equipped with an addition calibration storage facility.

2.2.2.3.2. Reagents.

All analytical solutions were prepared using chemicals of Analar quality and grade A glassware.

Sodium hydroxide solution (5.00 mol dm^{-3}) for alkali fusion extractions of fluoride and chromium (section 2.2.2.3.3.) consisted of sodium hydroxide pellets (200 g) dissolved in distilled water and diluted to 1 dm^3.

Total ionic strength adjustment buffer (TISAB) for fluoride determinations (section 2.2.2.3.4.) consisted of glacial acetic acid (58 cm^3) and sodium citrate (12 g) dissolved in 300 cm^3 of distilled water, adjusted to pH 5.2 using sodium hydroxide solution (5.00 mol dm^{-3}) and made up to 1 dm^3. A standard fluoride solution (0.20 mol dm^{-3}) consisted of sodium fluoride (8.3980 g) dissolved in distilled water and made up to 1 dm^3. Fluoride solutions of 0.02000, 0.00200, 0.00020 and 0.00002 mol dm^{-3} were produced by sequential dilutions of the original standard.

Acidifying solution for chromium determinations (section 2.2.2.3.5.) consisted of 200 cm^3 of sulphuric acid (2.50 mol dm^{-3}) and 100 cm^3 of sodium sulphate solution (30 g/dm^3) made up to 1 dm^3. Sodium dichromate (2.8660 g) was dissolved and made up to 1 dm^3 with distilled water to provide a standard chromium solution (1000 ug/cm^3). A chromium solution (25 ug/cm^3) was prepared by dilution of the standard solution.

2.2.2.3.3. Fluoride and chromium extraction method.

Approximately 0.25 g of each wood sample (section 2.2.2.2.2.) was weighed into each of 2 nickel crucibles (70 cm^3 capacity). Ten cm^3 of sodium hydroxide solution (5.00 mol dm^{-3}) was added to each crucible which was heated for approximately 40 minutes on a hot plate set at 150°C. The dried sample was placed over a Meker (Amalmajor) gas burner and brought to red heat over a 5 minute period using a low flame initially to avoid combustion. Red heat was maintained on full flame for 30 seconds or until the fused mixture was a uniform red colour. The crucible was removed to allow contents to cool and solidify. The solidified sample was broken up by gently heating and stirring after addition of 10 cm^3 of distilled water. Sodium peroxide (1 g) was quickly added with vigorous stirring. The mixture was allowed to cool and 6 cm^3 of concentrated hydrochloric acid was slowly added with stirring to adjust the pH to 8-9 (monitored with the use of pH paper). The contents of the crucible were left to cool then washed through Whatman No.4 filter paper into a 100 cm^3 volumetric flask and made up to volume with distilled water. Each group of wood samples carried through the extraction procedure was accompanied by 2 crucibles containing reagents only.

Approximately 0.25 g of each wood sample was weighed onto a watchglass and held overnight in an oven at 105°C. The samples were cooled in a desiccator and dry weights recorded. The crucible samples were corrected for dry weight as follows:

$$\frac{\text{Dry weight of oven sample (g)}}{\text{\textit{Wet} weight of oven sample (g)}} \times \text{\textit{Wet} weight of crucible sample (g)} = \mathbf{W}$$

2.2.2.3.4. Fluoride determination.

Measurement of sample solutions.

Twenty five cm^3 of each sample solution (section 2.2.2.3.3.) was added to 25 cm^3 of TISAB in a polyethylene beaker. The fluoride and reference electrodes were placed in the solution which was carefully swirled by hand for 1 minute and left to settle for a further 2 minutes, when a potential measurement was recorded in millivolts (mV). Five cm^3 of a standard fluoride solution known, by reference to a standard curve of mV versus molarity of the standard solutions (section 2.2.2.3.2), to be 5-10 times as concentrated as the sample solution, was added to the sample solution, mixed as before and a final potential was recorded. The fluoride concentration ratio (Q) corresponding to the change in mV potential between readings (AE) was found by reference to a known addition table of values supplied by Russell pH Limited (table 2.2.1). To determine the original total sample concentration of fluoride, the concentration ratio (Q) was multiplied by the concentration of added standard.

e.g. For a hypothetical sample of potential measurement -11.4 mV reduced to -26.8 mV by addition of 5 cm^3 of a fluoride solution (0.002 mol dm^{-3}), AE = 15.4 mV and the concentration ratio (Q) by reference to the table of values (table 2.2.1) is 0.0997.

Therefore $0.0997 \times 0.002 = 1.994 \times 10^{-4}$ mol dm^{-3}.

As half the original solution in the beaker was TISAB, the molarity of the sample solution was 3.988×10^{-4}, therefore 100 cm^3 contained 3.988×10^{-5} Moles of fluorine (molecular weight 19.00). Therefore, the fluoride content (ugF) of the sample solution (100 cm^3) was $39.88 \times 19.00 = 757.72$ ug.

Table 2.2.1. Known addition table showing fluoride concentration ratio values (Q) corresponding to the change in mV potential (AE) of the original sample solution after a 10 % volume addition of a known fluoride standard.

AE	Q	AE	Q	AE	Q	AE	Q	AE	Q	AE	Q	AE	Q	AE	Q
5	0.297	7.5	0.212	10	0.16	15	0.103	20	0.0716	25	0.0523	30	0.0394	35	0.0304
5.1	0.293	7.6	0.209	10.2	0.157	15.2	0.1013	20.2	0.0707	25.2	0.0517	30.2	0.039	35.2	0.0289
5.2	0.288	7.7	0.207	10.4	0.154	**15.4**	**0.0997**	20.4	0.0698	25.4	0.0511	30.4	0.0386	35.4	0.0275
5.3	0.284	7.8	0.204	10.6	0.151	15.6	0.0982	20.6	0.0689	25.6	0.0505	30.6	0.0382	35.6	0.0261
5.4	0.28	7.9	0.202	10.8	0.148	15.8	0.0967	20.8	0.068	25.8	0.0499	30.8	0.0378	35.8	0.0249
5.5	0.276	8	0.199	11	0.145	16	0.0952	21	0.0671	26	0.0494	31	0.0374	40	0.0237
5.6	0.272	8.1	0.197	11.2	0.143	16.2	0.0938	21.2	0.0662	26.2	0.0488	31.2	0.037	41	0.0226
5.7	0.268	8.2	0.195	11.4	0.14	16.4	0.0924	21.4	0.0654	26.4	0.0482	31.4	0.0366	42	0.0216
5.8	0.264	8.3	0.193	11.6	0.137	16.6	0.091	21.6	0.0645	26.6	0.0477	31.6	0.0362	43	0.0206
5.9	0.26	8.4	0.19	11.8	0.135	16.8	0.0897	21.8	0.0637	26.8	0.0471	31.8	0.0358	44	0.0196
6	0.257	8.5	0.188	12	0.133	17	0.0884	22	0.0629	27	0.0466	32	0.0354	45	0.0187
6.1	0.253	8.6	0.186	12.2	0.13	17.2	0.0871	22.2	0.0621	27.2	0.0461	32.2	0.0351	46	0.0179
6.2	0.25	8.7	0.184	12.4	0.128	17.4	0.0858	22.4	0.0613	27.4	0.0456	32.4	0.0347	47	0.0171
6.3	0.247	8.8	0.182	12.6	0.126	17.6	0.0846	22.6	0.0606	27.6	0.045	32.6	0.0343	48	0.0163
6.4	0.243	8.9	0.18	12.8	0.123	17.8	0.0834	22.8	0.0598	27.8	0.0445	32.8	0.034	49	0.0156
6.5	0.24	9	0.178	13	0.121	18	0.0822	23	0.0591	28	0.044	33	0.0336	50	0.0149
6.6	0.237	9.1	0.176	13.2	0.119	18.2	0.0811	23.2	0.0584	28.2	0.0435	33.2	0.0333	51	0.0143
6.7	0.234	9.2	0.174	13.4	0.117	18.4	0.0799	23.4	0.0576	28.4	0.0431	33.4	0.0329	52	0.0137
6.8	0.231	9.3	0.173	13.6	0.115	18.6	0.0788	23.6	0.0569	28.6	0.0426	33.6	0.0326	53	0.0131
6.9	0.228	9.4	0.171	13.8	0.113	18.8	0.0777	23.8	0.0563	28.8	0.0421	33.8	0.0323	54	0.0125
7	0.225	9.5	0.169	14	0.112	19	0.0767	24	0.0556	29	0.0417	34	0.0319	55	0.012
7.1	0.222	9.6	0.167	14.2	0.11	19.2	0.0767	24.2	0.0549	29.2	0.0412	34.2	0.0316	56	0.0115
7.2	0.219	9.7	0.165	14.4	0.108	19.4	0.0746	24.4	0.0543	29.4	0.0408	34.4	0.0313	57	0.011
7.3	0.217	9.8	0.164	14.6	0.106	19.6	0.0736	24.6	0.0536	29.6	0.0403	34.6	0.031	58	0.0105
7.4	0.214	9.9	0.162	14.8	0.105	19.8	0.0726	24.8	0.053	29.8	0.0399	34.8	0.0307	59	0.0101

Calculation of the fluoride content of wood samples.

The mean fluoride content (ug) of the reagent blank solutions was subtracted from the fluoride content of each sample solution (ug) and the concentration of fluoride in the original wood sample (% w/w) was calculated by the formula

$$\% \text{ w/w} = \frac{\text{F (ug of fluoride in 100 cm}^3\text{ sample solution)}}{\text{W (corrected dry weight of sample in ug)}} \times 100$$

2.2.2.3.5. Chromium determination.

Atomic absorption spectrophotometer - conditions of operation.

The instrument (section 2.2.2.3.1.) was fitted with a chromium hollow cathode lamp and standard operating conditions for analysis of the element were maintained; a wavelength of 357.9 nm, a yellow reducing flame fuelled by acetylene with air as oxidant. The sensitivity of the instrument was maximised using a machine standard chromium solution (2 ug/cm^3) to achieve a stable absorbance of at least 0.100 compared with zero absorbance for distilled water.

Preparation of sample solutions.

Sample solutions (section 2.2.2.3.3.) were initially separated into concentration groups of those likely to contain high and low levels of total chromium. The former group for instance, included sample solutions obtained from wood samples 1 and 4 which included the preservative injection site (section 2.2.2.2.2. and figure 2.2.1).

An aliquot of 1 sample solution from each concentration group was pipetted into each of 3 volumetric flasks of 25 cm^3 (A, B and C) each containing 1 cm^3 of acidifying solution. Zero, 1 and 2 cm^3 of a chromium solution (25 ug/cm^3) was pipetted into flasks A, B and C respectively and each made up to volume with distilled water. No flask contained a chromium concentration greater than 5 ug/cm^3, the limit of the direct linear relationship between the concentration of chromium ions in solution and their absorbance at 357.9 nm. Single flasks of the remaining sample solutions from each concentration group were prepared identically to flask A. A reagent blank solution for each group was prepared likewise.

This procedure provided duplicate flasks for each original wood sample (section 2.2.2.2.2.) and ensured that each sample solution of similar concentration was of identical dilution and provided with a similarly diluted reagent blank solution. Each concentration group included 2 solutions, in flasks B and C, containing 1 ug/cm^3 and 2 ug/cm^3 concentrations of chromium respectively, in addition to the unknown sample concentration, within an equivalent chemical matrix to the sample solutions.

Measurement of sample solutions.

Each concentration group was measured separately. The absorbance of the reagent blank solution was used to re-zero the addition calibration program of the instrument. The absorbance readings of flasks A, B and C were displayed as a calibration curve calculated and stored by the instrument. The concentration of chromium in flask A, the initial sample solution, was calculated and given in ug/cm^3. The remaining sample solutions were read and the concentrations of chromium present automatically calculated from the curve. The reading recorded for each flask was the mean of two consecutive five second readings. The accuracy of measurements was checked by regular readings of a number of flasks containing reagent blank solutions with known additions of chromium.

Calculation of the chromium content of wood samples.

The chromium content of each original wood sample (% w/w) was calculated from each duplicate flask concentration, after subtraction of the reagent blank concentration, using the formula

$$\% \text{ w/w} = \frac{C \times 2500 \times 100}{V \times W}$$

where

C = concentration of chromium in sample flask (ug/cm^3).

2500 = volume of flask x volume of original sample solution (cm^3).

V = volume of aliquot (cm^3).

W = dry weight of wood sample (ug).

2.2.3. The effects of Rentex remedial treatment on some wood pole inhabitant micro-organisms.

2.2.3.1 Summary.

Wood cores were removed from creosoted distribution poles up to 16 months after Rentex remedial treatment and, over the same period, from creosoted distribution poles which had received no remedial treatment. To examine any effects of the preservative on pole inhabitant micro-organisms, the uncreosoted portion of wood cores from remedially treated and untreated poles was incubated on nutrient agar to enable growth and identification of the microbial species present.

2.2.3.2. Field operations.

2.2.3.2.1. Excavation and measurement of distribution poles.

In December 1989, 240 creosoted distribution poles, in service since 1958 and situated at Glen Clova in the east of Scotland, were excavated to a depth of 0.5-0.75 m and pole circumferences recorded at the groundline. The height of each pole above the groundline and depth below it was recorded from the pole suppliers identification stamp present on each pole surface. Due to possible effects on pole bending strength, the use of these poles was permitted for core sampling procedures (sections 2.2.3.2.2. and 2.2.3.2.6) on the understanding that a maximum of 7 wood cores could be removed from each pole.

2.2.3.2.2. Initial core samples.

Four wood cores of bore 0.5 cm and measuring approximately half the pole diameter in length were recovered from each of the 240 poles using a *Mattson* augur. The augur was dipped in industrial alcohol and flame sterilised between each sample. Cores were removed, in a vertical line, at 35 cm and 17.5 cm above the groundline, 17.5 cm below the groundline and at the groundline itself with the augur positioned at right angles to the pole surface. Each core was immediately inserted aseptically into a labelled sterile screw top test tube to be used for later isolation and identification of inhabitant micro-organisms (section 2.2.3.3.1.) and for creosote depth measurements (section 2.2.3.3.2.) in the laboratory.

2.2.3.2.3. Measurement of pole moisture contents.

The % moisture content of each pole was recorded using a *Protimeter* conductivity moisture meter (section 2.2.2.2.2.) attached to an elongated probe developed by the Midlands Electricity Board. The probe was fully inserted into the groundline borehole and pressed firmly to the pole interior thereby measuring conductivity across 2 contact points on the end of the probe and hence % moisture content. The core sample boreholes (section 2.2.3.2.2.) were plugged with softwood dowels, which, for poles awaiting preservative treatment (section 2.2.3.2.4.), had been pressure impregnated with a 5% solution of Rentex.

2.2.3.2.4. Pole treatment.

After initial core removal and measurement of pole moisture content every alternate distribution pole of the 240 excavated was Rentex treated according to the 'Cobra' method (sections 1.6.2. and 2.1.2.). This provided 120 treated and 120 untreated poles, each treated pole flanked by 2 untreated control poles and vice versa. Treated and control poles were therefore equally represented at any particular site within the larger Glen Clova area. Preservative injections ensured that the sealed boreholes (section 2.2.3.2.3.) were

positioned centrally within a diamond pattern of 4 injections characteristic of the treatment method (section 2.1.2.). After preservative treatment or sealing of boreholes, for control poles, soil excavations adjacent to poles were refilled.

2.2.3.2.5. Pole ranking system.

To facilitate monitoring of the effects of remedial treatment on the microbial populations of distribution poles (section 2.2.3.2.6.) each group of 120 remedially treated and 120 control poles were ranked from lowest to highest moisture contents based on the initially recorded pole moisture contents (section 2.2.3.2.3.), as this pole parameter was likely to have a major influence on preservative efficacy (sections 2.1.2. and 2.1.3.). Each group (treated and control) was split into 20 smaller groups of 6 (numbered 1-6) starting with poles of lowest moisture content and proceeding through to poles of highest moisture content. This grouping system provided 20 poles numbered 1, 20 numbered 2 and so on through to 20 poles numbered 6, for each group of 120 control and treated poles.

2.2.3.2.6. Final core samples.

At 1 month after remedial treatment 20 treated and 20 control poles, each numbered 1 according to the ranking system, were excavated. The aluminium sheath fitted to each treated pole was carefully removed. Three core samples were removed as before, from each pole next to the original 3 lower sampling positions from the middle of the adjacent diamond pattern of injections for isolation of micro-organisms (section 2.2.3.3.1.). Pole moisture contents were recorded at the groundline to confirm the original moisture status of each pole and core boreholes were plugged as before (section 2.2.3.2.3.). Aluminium sheaths were refitted to treated poles and excavations refilled.

At 3, 6 and 16 months after remedial treatment a separate group of 20 treated and 20 control poles, each numbered 2, 3, and 4 respectively were similarly excavated and

sampled. Thus, a total of 80 treated and 80 control poles were core sampled a second time ensuring that core samples from treated and control poles of high and low moisture content were equitably spread throughout the sample population at each collection period.

2.2.3.3. Laboratory procedures.

2.2.3.3.1. Isolation/identification of micro-organisms from core samples.

The creosoted portion of each core recovered from each pole (section 2.2.3.2.2. and 2.2.3.2.6.), excepting that recovered from 35 cm above the groundline (section 2.2.3.2.2.), was discarded and the portion remaining aseptically transferred into a labelled petri dish containing 3 % malt extract agar. The petri dishes were incubated in the dark for 1 month at 25°C to allow growth of any micro-organisms present.

Growth of bacteria was recorded as such. Suspected moulds were sub-cultured onto 3% malt extract agar, incubated at 25°C and after sufficient growth, the major species were differentiated by reference to standard texts for identification of fungi (Barnett, 1958; Wang and Zabel, 1990). The majority of moulds were found to be strains of ***Cladosporium resinae*** and ***Trichoderma*** species, and the presence of these moulds on wood cores was recorded separately and as part of a larger group termed 'moulds'. Suspected basidiomycetes were sub-cultured onto 3% malt extract agar, containing 4 ppm Benomyl to prevent the growth of moulds (see section 2.2.1.5.). Growth on Benomyl agar and the presence of mycelial clamp connections identified these fungi as basidiomycetes. Strains of the basidiomycete ***Neolentinus lepideus***, differentiated by morphological and cultural characterisation together with cross referencing against stock cultures of this fungus, were the most commonly isolated. The identity of ***N. lepideus*** isolates was confirmed using an identification key (Noble, 1964) and their presence on wood cores recorded.

2.2.3.3.2. Creosote depth measurements.

The core recovered from 35 cm above the groundline of each pole section (section 2.2.3.2.2.) was used to determine creosote depth as a percentage of pole radius, calculated from the recorded circumference of each pole (section 2.2.3.2.1.), and for measurement of any uncreosoted sapwood. Each core was sprayed with a 1:1 mixture of sodium nitrite solution (10 g sodium nitrite dissolved in 100 cm^3 distilled water) and o-anisidine solution [0.5 g o-anisidine dissolved in 100 cm^3 of dilute hydrochloric acid (1.75 cm^3 of concentrated hydrochloric acid made up to 100 cm^3 with distilled water)] (Stalker, 1971). The solution stains the heartwood (*Pinus* species) a deep red colour, by reaction with phenolic compounds, thus identifying uncreosoted sapwood as a pale yellow unstained area adjacent to the creosoted portion of the core. The use of this procedure indicated that none of the poles in this study contained areas of uncreosoted sapwood.

2.2.4. Fluoride and chromium concentrations of wood cores recovered from remedially treated distribution poles.

2.2.4.1. Summary.

Of the 120 remedially treated distribution poles at the Glen Clova field site (section 2.2.3.2.1.), 11 were selected as having moisture contents ranging across the spectrum of those recorded at this field site (section 2.2.3.2.3.) and having had no final set of cores removed for isolation studies. Core samples were removed from these poles 18 months after remedial treatment and chemically analysed to determine the loadings of fluoride and chromium throughout the groundline region.

2.2.4.2. Core sampling and pole measurements.

Eleven poles had their aluminium sheaths removed and were excavated to a depth of 0.5-0.75 m. Three core samples of bore 0.5 cm were recovered using a *Mattson* augur from around the pole at 3 heights, i.e. at 17.5 cm above and below the groundline and at the groundline. Care was taken to avoid sites of preservative injection. The pole parameters of moisture content, pole height, pole depth and percentage creosote depth were recorded, core boreholes sealed, aluminium sheaths restored and excavations refilled as for other poles at this site (section 2.2.3.).

2.2.4.3. Preparation and chemical analysis of core samples.

The creosoted portion of each core was discarded and the cores from each pole combined and ground in a hammer mill prior to duplicate chemical analysis for fluoride and

chromium content (% w/w) as previously (section 2.2.2.3.).

2.2.5. Measurement of fluoride and chromium concentrations in soils adjacent to Rentex treated creosoted field poles.

2.2.5.1. Summary.

Soil samples, recovered from the vicinity of Rentex treated distribution poles up to 1 year after preservative treatment, were analysed to identify any significant increase in the concentrations of the preservative constituents fluoride and chromium in soils adjacent to treated timber compared to background soil levels.

2.2.5.2. Pole selection.

In December 1989, 14 distribution poles from 120 undergoing remedial injection with Rentex preservative at the Glen Clova field site (section 2.2.3.2.4.) were selected on the basis of groundline measurements of pole moisture contents (section 2.2.3.2.3.) to provide 7 poles of high and 7 of low moisture content. Poles of low recorded moisture content were situated in elevated hill positions in well drained sandy loam soils whereas the former group of poles were found in low lying situations in sandy clay loam and silty clay soils, which suffered periodic surface flooding.

2.2.5.3. Soil sampling.

A steel wood drill with a 15 mm diameter head, modified by welding to a steel stalk, proved the most practical and least destructive tool for sampling of the stony soils adjacent to poles. The stony nature of the soil dictated sample depth. The removal of a series of depth samples from each borehole was prevented by difficulties in extracting comparable

soil volumes from different depths and different sites. Sampling was carried out by twisting the drill head of the sampler into the soil until buried and pulling gently upwards, the head retaining soil on its spiral edge, which was removed by shaking into a plastic sample bag. Sampling within the same borehole continued till a depth of 60 cm was reached, measured by a gradation mark on the steel stalk of the sampler. Each complete sample was sealed in a plastic bag.

Soil samples were collected from 6 cm and 25 cm downslope of each pole at 1 week, then 1, 6 and 12 months after remedial treatment of the poles. A background soil sample from approximately 50 m upslope of each pole was also recovered at these times.

2.2.5.4. Preparation and chemical analysis of samples.

The moist soil samples were placed on plastic petri dishes in the laboratory to air dry, ground to pass a 2 mm mesh stainless steel sieve then ground to fine particle size and sealed in plastic bags. Duplicate soil chemical analysis for fluoride and chromium content was carried out using representative 0.25 g samples as for wood samples (section 2.2.2.3.) and expressed in micrograms (ug) of fluoride and chromium per gram (g) oven dry weight of soil (section 2.3.5.).

2.2.6. Measurement of fluoride and chromium losses from Rentex treated creosoted pole sections after extended periods of field exposure.

2.2.6.1. Introduction.

Two field sites containing Rentex treated creosoted Scots pine pole sections were established to evaluate the preservative treatment. One, situated near Oban in the west of Scotland was established in July 1987 by Cobra (Wood Treatment) Limited under the supervision of the Wood Research Group of Dundee Institute of Technology (Bruce and King, 1989). Though the unspecified Rentex formulation used at this field site was not identical to that used in the present study, it did contain sodium fluoride and ammonium bifluoride and was applied by the same injection process. Consequently, this field site was thought worthy of further study. The other field site at Tealing in the east of Scotland was established in November 1989 as part of the present study (see section 2.2.2.2.1.). Pole sections removed from the Oban site had been subjected to 4.25 years field exposure, while those removed from the Tealing site had been exposed for up to 2 years. All pole sections were analysed for fluorie and chromium content to identify any loss of these preservative constituents.

2.2.6.2. Pole section preparation and treatment.

All pole sections were cut from the mid to upper portions of aged creosoted distribution poles, their tops were tapered and bitumen coated to facilitate rainfall run-off. At the Oban site, pole sections were approximately 4 m in length and at Tealing 6 m, and all were erected to a depth of approximately 1 m. All the pole sections received groundline treatment with Rentex by the 'Cobra' injection process (sections 1.6.2. and 2.1.2.), and the circumference of each pole section and the number of injections applied to the Tealing pole

sections was recorded.

2.2.6.3. Recovery of pole sections.

Seven pole sections removed from the Tealing field site immediately after treatment (section 2.2.2.2.2.) were stored indoors and unwrapped for 2 years. A further 7 pole sections were uplifted from this site after 2 years field exposure and 5 pole sections from the Oban site after 4.25 years. That portion of each field exposed pole section above the aluminium sheath of the remedially treated zone was cut and discarded on site. The moisture content of each pole section was recorded at the cut surface (section 2.2.2.2.2.) before removal from each site. The moisture contents of the 7 unexposed pole sections from the Tealing site were similarly recorded after 2 years storage.

2.2.6.4. Sampling of pole sections.

The aluminium sheath encompassing the upper half of the injected area was removed. The entire treated area, 35 cm above and below the groundline position, was cut away from each pole section using a *Stihl* chainsaw.

From the top of each treated zone, a 1-2 cm deep disc was cut. Each disc diameter was measured, creosote depth recorded as a percentage of pole section radius and visible preservative injections counted. Each disc was quartered and the exact dimensions of 1 quarter measured for an accurate calculation of volume. The quarters were placed in an oven set at 105°C for 2-3 days, cooled in a dessicator and dry constant weights recorded. From the volume (cm^3) and dry weight (g) of each pole section quarter, the density (g/cm^3) of each remedially treated creosoted pole section was calculated.

Using a *Stihl* chainsaw the remaining treated area of each pole section was severed at an angle of 45° to the horizontal at the groundline position and 17.5 cm above and below it. The sawdust samples collected from each cut were mixed thoroughly, spread to a depth of approximately 2 cm and quartered. One quarter was retained, mixed, spread again and quartered once more. One quarter of approximately 25 g in weight was retained and sealed in a plastic bag. The sampling of all pole sections provided 57 sawdust samples, i.e. 19 pole sections each sampled at 3 heights, from 3 different pole groups.

2.2.6.5. Chemical analysis of sawdust samples.

Duplicate chemical analysis of each sawdust sample for fluoride and chromium content was carried out as for wood samples for evaluation of preservative distribution in treated timber (section 2.2.2.3.).

2.2.7. Analysis of soils adjacent to Rentex treated creosoted pole sections for fluoride and chromium content.

2.2.7.1. Summary.

To confirm any leaching losses of fluoride and chromium preservative constituents from field exposed remedially treated pole sections (section 2.2.6.), soil samples were collected for chemical analysis simultaneously with removal of pole sections from each field site (section 2.2.6.3.).

2.2.7.2. Soil sampling regimes.

The topographical and soil conditions characteristic of each field site determined the soil sampling regime employed. At Tealing, a level site, the soil was a free draining sandy loam supporting a grass sward. Oban was a sloping site of the same soil type, which, due to overworking, suffered a degree of waterlogging and sparse plant cover.

At both field sites, soil samples were collected at each pole section and a background sample was collected at least 20 m from each pole section. A further soil sample was collected 25 cm downslope of each pole section at Oban, site conditions here encouraging lateral flow of drainage water. Samples at each pole section consisted of soil adhering to the surface of the lower 35 cm of the remedially treated zone which was carefully brushed into plastic sample bags as each pole section was uplifted (section 2.2.6.3.). Background samples were collected by uplifting a 35 cm deep turf and slowly scoring its face with a stainless steel spatula, retaining the loosened soil in a sealed plastic sample bag.

2.2.7.3. Preparation and chemical analysis of soil samples.

Each soil sample was air dried, ground and a duplicate chemical analysis for fluoride and chromium content carried out as for soils adjacent to remedially treated distribution poles (section 2.2.5.4.).

2.3. RESULTS.

2.3.1. General layout of results sections.

The results for each of the studies detailed in sections 2.2.1., 2.2.2., 2.2.3., 2.2.4., 2.2.5., 2.2.6. and 2.2.7. are presented in tables and/or figures at the beginning of sections 2.3.2., 2.3.3., 2.3.4., 2.3.5., 2.3.6., 2.3.7. and 2.3.8. respectively, and are followed by descriptions of results. All statistical analyses were carried out using the MINITAB statistical computer package (Copyright 1992 Minitab Inc.).

2.3.2. Toxicity of Rentex impregnated wood to selected fungi.

2.3.2.1. Results tables.

The substantial decay in untreated Scots pine wood blocks caused by basidiomycetes is indicated by the loss in mass of *virulence control* sapwood and heartwood blocks after exposure to ***Neolentinus lepideus*** (pole isolate 4), ***N. lepideus*** (BAM 20), ***Coniophora puteana*** (BAM 15) and ***Coriolus versicolor*** (FPRL 28B) shown in table 2.3.2.1. This table also indicates the prolific mould growth found on untreated sapwood blocks after exposure to ***Cladosporium resinae*** (pole isolate) and ***Trichoderma polysporum*** (IMI 206039).

Weight losses of preservative impregnated and control sapwood blocks exposed to pure cultures of the basidiomycetes ***Neolentinus lepideus*** (pole isolate 4), ***N. lepideus*** (BAM 20), ***Coniophora puteana*** (BAM 15) and ***Coriolus versicolor*** (FPRL 28B) are presented in tables 2.3.2.2, 2.3.2.4, 2.3.2.6 and 2.3.2.8 respectively.

Weight losses of preservative impregnated and control heartwood blocks exposed to pure cultures of ***N. lepideus*** (pole isolate 4), ***N. lepideus*** (BAM 20) and ***C. puteana*** (BAM 15) are presented in tables 2.3.2.3, 2.3.2.5 and 2.3.2.7 respectively.

Scores for growth of ***Cladosporium resinae*** (pole isolate) and ***Trichoderma polysporum*** (IMI 206039) over preservative impregnated and control sapwood blocks are presented in tables 2.3.2.9 and 2.3.2.10 respectively.

The preservative concentrations which provided a protective toxic threshold to basidiomycete decay and mould growth in impregnated sapwood and heartwood blocks, based on the finding in tables 2.3.2.2 - 2.3.2.10 (see section 2.2.1.2.6), are presented in table 2.3.2.11, as the treatment concentration of Rentex, retention of the product (kg/m^3), and the concentration of Rentex and fluoride in the treated blocks (% w/w).

2.3.2.2. Virulence of fungal strains.

The loss in mass of untreated *virulence control* Scots pine sapwood and heartwood blocks after exposure to basidiomycetes (table 2.3.2.1) shows the degree of decay caused by these micro-organisms and the severity of the exposure to which Rentex impregnated wood block specimens were subjected. Comparison of the mean loss in mass of either sapwood or heartwood *virulence control* blocks after exposure to different basidiomycete strains (table 2.3.2.1) indicates the similarity in virulence of these basidiomycetes. The mean percentage loss in mass of *virulence control* heartwood blocks and similarly exposed sapwood blocks (table 2.3.2.1) were not markedly different.

Untreated *virulence control* sapwood blocks exposed to the moulds ***T. polysporum*** and ***C. resinae*** were generally completely overgrown (table 2.3.2.1) indicating that any restriction of mould growth on the treated specimens (tables 2.3.2.9 and 2.3.2.10) was not

Table 2.3.2.1. Loss in mass/Growth scores for Scots pine sapwood and heartwood 'virulence control' blocks after 16 weeks exposure to selected basidiomycetes (B) and moulds (M) (standard deviations for means of 6 in parenthesis).

Wood block type	Basidiomycete or Mould (B/M)		Corrected mass loss (%) or score (0-4)	Mean mass loss (%) or score (0-4)
Sapwood	*N. lepideus, 4*	B	35 37 23 38 28 29	31.4 (5.6)
	N. lepideus, 20	B	31 35 34 29 36 33	33.0 (2.4)
	C. puteana	B	35 37 23 38 28 29	31.7 (5.4)
	C. versicolor	B	27 25 29 35 55 25	32.7 (10.5)
	T. polysporum	M	4 4 4 4 4 4 *	4.0 (0.00)
	C. resinae	M	3 4 4 4 3 4 *	3.7 (0.47)
Heartwood	*N. lepideus, 4*	B	43 36 33 37 42 28	36.4 (5.6)
	N. lepideus, 20	B	29 30 28 32 29 27	29.2 (1.6)
	C. puteana	B	35 38 35 32 36 26	33.7 (3.9)

** The score of 4 represents 100 % mould growth over the block.

Table 2.3.2.2. Mean percentage weight losses of Rentex treated and control sapwood blocks exposed together for 16 weeks to N. lepideus, pole isolate 4 (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	Mean retention of Rentex: kg/m^3	Corrected loss in mass (%)	Mean loss in series (%)	Mean loss in mass of controls (%)
0	* 14.65 (0.93)	* 0	0	26.8 22.2 29.7 23.5	* 25.6 (2.92)	** 18.3 (4.9)
0.05	13.35 (0.98)	775 (50.2)	0.409 (0.03)	32.7 16 18.8 16.6	21 (6.82)	29.1 (2.1)
0.1	13.2 (0.83)	1550 (50)	0.82 (0.03)	12.9 16.2 11.1 12.7	13.2 (1.85)	23.1 (1.3)
0.25	13.7 (0.82)	3725 (303)	1.96 (0.16)	2.2 1.9 0.8 1.4	1.6 (0.53)	34.8 (6.3)
0.5	12.2 (0.63)	8300 (380)	4.38 (0.20)	0 0 0.8 0	0.2 (0.35)	20.1 (5.5)
1	13.2 (1.14)	13900 (724)	7.33 (0.38)	0 0 0 0.1	0 (0.04)	10.2 (9.3)
2	13.5 (0.52)	32975 (1215)	17.39 (0.64)	0.2 0.2 0.2 0.7	0.3 (0.22)	14.6 (3.7)

Table 2.3.2.3. Mean percentage weight losses of Rentex treated and control heartwood blocks exposed together for 16 weeks to N. lepideus, pole isolate 4 (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Corrected loss in mass (%)	Mean loss in series (%)	Mean loss in mass of controls (%)
0	* 13.6 (0.67)	* 0	0	20.7 25.3 28.6 21.2	* 24 (3.2)	** 14.3 (8.8)
0.05	12.5 (0.98)	847 (54.5)	0.34 (0.02)	31.9 21 14.2 13.8	20.2 (7.3)	13.2 (1.6)
0.1	13.2 (0.56)	1400 (70.7)	0.56 (0.03)	14.6 8.8 12.7 11.1	11.8 (2.1)	24.3 (3.0)
0.25	12.25 (1.05)	3700 (430)	1.47 (0.17)	4.2 5.3 2.2 1.7	3.4 (1.4)	13 (1.9)
0.5	13 (0.66)	8260 (738)	3.29 (0.29)	3.5 2.8 1.3 1.7	2.3 (0.9)	21 (2.8)
1	12.9 (0.6)	17000 (4243)	6.78 (1.67)	0.2 1.4 0.2 5.7	1.9 (2.3)	21.3 (5.5)
2	13.6 (0.53)	33050 (5650)	13.17 (2.23)	0.2 0.2 0.2 0.7	0.2 (0.00)	14.6 (3.7)

Table 2.3.2.4. Mean percentage weight losses of Rentex treated and control sapwood blocks exposed together for 16 weeks to N. lepideus, BAM 20 (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Corrected loss in mass (%)	Mean loss in series (%)	Mean loss in mass of controls (%)
0	* 12.5 (0.78)	* 0	0	24.9 15.6 8.1 42	* 22.6 (12.7)	** 17.7 (3.6)
0.05	12.9 (0.70)	725 (38.4)	0.38 (0.02)	18.9 17.6 14 12.4	15.7 (2.6)	17.4 (1.8)
0.1	13 (0.58)	1575 (82.9)	0.83 (0.04)	8 9.6 6.8 8	8.1 (0.99)	16.8 (2.4)
0.25	11.9 (0.75)	3650 (296)	1.92 (0.16)	3.2 1.9 1.4 1.7	2 (0.69)	19.5 (2.7)
0.5	13.2 (0.23)	8150 (602)	4.30 (0.32)	0.8 0 0.2 0	0.25 (0.33)	15.2 (1.9)
1	13.2 (0.78)	14725 (742)	7.77 (0.39)	0.4 0 0.2 0	0.2 (0.18)	12.7 (2.8)
2	12.6 (0.76)	30750 (260)	16.22 (0.14)	0.2 0.2 0.2 0.2	0.2 (0.0)	14.1 (8.7)

Table 2.3.2.5. Mean percentage weight losses of Rentex treated and control heartwood blocks exposed together for 16 weeks to N. lepideus, BAM 20 (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Corrected loss in mass (%)	Mean loss in series (%)	Mean loss in mass of controls (%)
0	* 13.4 (0.77)	* 0	0	33.2 29.6 42.7 30.3	* 35.5 (4.8)	** 28.9 (2.8)
0.05	12.9 (1.15)	730 (43.6)	0.29 (0.02)	35.1 29.7 30.6 23.4	29.7 (4.2)	30.6 (4.8)
0.1	13.3 (0.91)	1550 (50.0)	0.62 (0.02)	21.2 18.8 11.6 25.1	19.2 (4.9)	19.1 (9.6)
0.25	13.1 (0.79)	3875 (334)	1.54 (0.13)	0.6 1.5 0.9 1.8	1.2 (0.5)	15.7 (3.7)
0.5	12.85 (1.07)	7825 (390)	3.12 (0.15)	0.8 0.6 1.2 0.8	0.8 (0.2)	21.6 (1.5)
1	13 (1.11)	16500 (2692)	6.58 (1.07)	0.5 0.4 0.2 0.3	0.3 (0.1)	24.6 (5.3)
2	12.8 (1.42)	36850 (3517)	14.69 (1.40)	0.2 0.2 0.2 0.2	0.2 (0.0)	18.9 (3.2)

Table 2.3.2.6. Mean percentage weight losses of Rentex treated and control sapwood blocks exposed together for 16 weeks to C. puteana, BAM 15 (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Corrected loss in mass (%)	Mean loss in series (%)	Mean loss in mass of controls (%)
0	* 13.8 (1.08)	* 0	0	62.8 25.4 55.8 24.7	* 42.2 (17.3)	** 39.3 (8.8)
0.05	13.25 (0.40)	740 (31.9)	0.39 (0.02)	50.3 66.8 51.2 57.3	56.4 (6.6)	57.6 (5.9)
0.1	13.12 (0.92)	1600 (70.7)	0.84 (0.04)	5.1 35.4 14.6 2.5	14.4 (12.9)	38.7 (9.7)
0.25	12.6 (1.22)	3450 (409)	1.82 (0.21)	8.3 17.2 4.6 7.7	9.4 (4.7)	54.9 (1.7)
0.5	13.65 (0.18)	7675 (277)	4.03 (0.15)	1.9 2.3 1.2 1.3	1.7 (0.4)	28.5 (3.9)
1	14.3 (0.8)	15550 (350)	8.2 (0.18)	0 0.5 lost lost	0.25 (0.2)	31.5 (7.3)
2	13.75 (0.54)	31550 (740)	16.64 (0.39)	0.2 0.2 0.2 0.2	0.2 (0.0)	35.9 (3.5)

Table 2.3.2.7. Mean percentage weight losses of Rentex treated and control heartwood blocks exposed together for 16 weeks to C. puteana, BAM 15 (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Corrected loss in mass (%)	Mean loss in series (%)	Mean loss in mass of controls (%)
0	* 13.1 (1.39)	* 0	0	22.3 22.4 28.3 27.7	* 25.2 (2.8)	** 29.6 (4.5)
0.05	13.2 (0.96)	740 (40.6)	0.29 (0.02)	42.2 41.9 50.4 10.1	36.1 (15.4)	40.1 (9.2)
0.1	13.1 (0.99)	1775 (356)	0.71 (0.14)	3.8 0 22.4 37.4	15.9 (15.0)	35.6 (9.8)
0.25	13.45 (0.41)	4350 (594)	1.73 (0.24)	2.3 4.5 12.7 6.5	6.5 (4.5)	49.3 (4.7)
0.5	13.2 (1.26)	8525 (991)	3.40 (0.39)	3.2 4.3 0 0	1.9 (1.9)	37.2 (5.0)
1	13.3 (0.86)	17000 (3240)	6.78 (1.29)	0.9 0.2 3.8 3.5	2.1 (1.6)	28.8 (14.2)
2	12.6 (0.88)	28000 (2362)	11.16 (0.94)	0.2 0.2 0.2 0.2	0.2 (0.0)	15.9 (11.9)

Table 2.3.2.8. Mean percentage weight losses of Rentex treated and control sapwood blocks exposed together for 16 weeks to C. versicolor, FPRL 28B (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Corrected loss in mass (%)	Mean loss in series (%)	Mean loss in mass of controls (%)
0	* 13.4 (0.65)	* 0	0	19.7 43.5 45.3 27.5	* 34 (10.8)	** 25.3 (2.5)
0.05	13 (0.70)	745 (47.2)	0.39 (0.02)	28.1 33.1 21.5 29.7	28.1 (4.2)	16.6 (7.4)
0.1	13.1 (0.51)	1550 (112)	0.82 (0.06)	9.8 12.6 8.2 11.4	10.5 (1.6)	18.3 (6.4)
0.25	12.9 (0.65)	3800 (70.7)	2.00 (0.04)	12.7 1.3 0.6 2.5	4.3 (4.9)	15.9 (2.7)
0.5	13.7 (0.88)	7900 (224)	4.17 (0.12)	0.2 0 0.1 0.8	0.3 (0.3)	35.7 (11.8)
1	12.9 (0.39)	15900 (534)	8.38 (0.24)	0 0.4 0 0	0.1 (0.2)	19.4 (2.8)
2	13.3 (0.84)	29900 (1678)	15.77 (0.88)	0.2 0.2 0.2 0.2	0.2 (0.0)	19.4 (4.3)

Table 2.3.2.9. Mean scores for growth of C. resinae, pole isolate, on Rentex treated and control sapwood blocks after 16 weeks exposure (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Score +++	Mean Score	Mean Score for controls
0	* 14.4 (1.75)	* 0	0	4 4 4 4	* 4 (0.00)	** 4 (0.0)
0.05	12.05 (0.61)	750 (18.7)	0.39 (0.01)	3 4 4 4	3.75 (0.43)	3 (0.0)
0.1	13.75 (0.94)	1525 (82.9)	0.80 (0.04)	3 3 3 3	3 (0.00)	3.5 (0.5)
0.25	13.1 (0.67)	3800 (255)	2.00 (0.13)	4 2 3 3	3 (0.70)	2.5 (0.5)
0.5	14 (0.38)	7625 (432)	4.02 (0.23)	1 2 2 2	1.75 (0.43)	3.5 (0.5)
1	12.8 (1.24)	14675 (1465)	7.74 (0.77)	2 2 1 1	1.5 (0.50)	3 (0.0)
2	13.1 (1.02)	31100 (1943)	16.40 (1.02)	0 0 0 1	0.25 (0.43)	3.5 (0.5)

+++ 0 = No mould growth, 1 = 25 % mould growth, 2 = 50 %, 3 = 75 % and 4 = 100

Table 2.3.2.10. Mean scores for growth of T. polysporum, IMI 206039, on Rentex treated and control sapwood blocks after 16 weeks exposure (standard deviations for means of 4* and 2** in parenthesis).

Rentex conc.s used (%)	Mean absorption by test specimens (g)	Mean retention of Rentex: ug/g	kg/m^3	Score +++	Mean Score	Mean Score for controls
0	* lost	* unknown		4 4 4 4	* 4 (0.00)	** 4 (0.0)
0.05	13.4 (1.07)	740 (39.3)	0.39 (0.02)	4 4 3 4	3.75 (0.43)	4 (0.0)
0.1	lost	unknown		4 3 3 4	3.5 (0.50)	4 (0.0)
0.25	12.8 (1.37)	3750 (166)	1.98 (0.06)	0 1 1 2	1 (0.71)	3 (0.0)
0.5	13.8 (0.11)	7350 (180)	3.88 (0.09)	1 1 1 0	0.75 (0.43)	1.5 (0.5)
1	12.3 (0.58)	15100 (717)	7.96 (0.38)	0 0 0 0	0 (0.00)	2 (0.0)
2	13 (1.12)	29200 (1964)	15.40 (1.04)	0 0 0 0	0 (0.00)	1 (0.0)

+++ 0 = No mould growth, 1 = 25 % mould growth, 2 = 50 %, 3 = 75 % and 4 = 100

Table 2.3.2.11. Toxic limits for Rentex to fungi, given as Rentex concentration in solvent, retention of product in wood blocks (kg/m^3) and as the concentration of Rentex and fluoride in blocks (% w/w).

Wood block type	Fungi	Rentex conc. in solvent (%)	Retention of Rentex (kg/m^3)	Rentex conc. in block (% w/w)	Fluoride conc. in block (% w/w)
Sapwood	*N. lepideus, 4*	0.10 - 0.25	0.82 - 1.96	0.16 - 0.37	0.030 - 0.071
	N. lepideus, 20	0.10 - 0.25	0.83 - 1.92	0.16 - 0.36	0.030 - 0.069
	C. puteana	0.25 - 0.50	1.82 - 4.03	0.35 - 0.72	0.066 - 0.140
	C. versicolor	0.25 - 0.50	2.00 - 4.17	0.38 - 0.79	0.072 - 0.150
	T. polysporum	1	7.96	1.51	0.29
	C. resinae	2	16.4	3.11	0.59
Heartwood	*N. lepideus, 4*	0.25 - 0.50	1.47 - 3.29	0.37 - 0.83	0.07 - 0.16
	N. lepideus, 20	0.10 - 0.25	0.62 - 1.54	0.16 - 0.39	0.03 - 0.07
	C. puteana	0.25 - 0.50	1.73 - 3.40	0.43 - 0.85	0.08 - 0.16

due to ambient experimental conditions.

2.3.2.3. Toxicity of treated wood to Basidiomycetes.

The mean loss in mass of preservative treated sapwood blocks, after exposure to each basidiomycete, was invariably lower than that of their respective controls and displayed a regular reduction in mean mass loss as preservative concentration increased (tables 2.3.2.2, 2.3.2.4, 2.3.2.6 and 2.3.2.8). Similarly, a reduction in mean mass loss of treated heartwood blocks was displayed as preservative concentration increased and these losses were always below control levels (tables 2.3.2.3, 2.3.2.5 and 2.3.2.7).

The preservative concentrations required to prevent decay of sapwood by the basidiomycetes (table 2.3.2.11) indicate the greater resistance of *C. puteana* and *C. versicolor* to the preservative compared with the 2 strains of *N. lepideus*. The concentrations of preservative required to prevent decay of heartwood blocks by *C. puteana* and *N. lepideus* (BAM 20) were the same as for sapwood, with *C. puteana* displaying the greater resistance to applied preservative (table 2.3.2.11). However, to prevent decay of heartwood due to *N. lepideus* (pole isolate 4), a higher preservative concentration than for sapwood was required.

The mean loss in mass of untreated sapwood control specimens after exposure to basidiomycetes in the same culture vessel as preservative treated specimens (tables 2.3.2.2, 2.3.2.4, 2.3.2.6 and 2.3.2.8) was, with the exception of those exposed to *C. puteana*, consistently lower and more irregular than that of respective untreated *virulence control* sapwood blocks in separate culture vessels (table 2.3.2.1). The likelihood that some transfer of preservative from treated blocks reduced the virulence of these basidiomycetes, is indicated by the tendency for the mean loss in mass of the sapwood control blocks to be lowest adjacent to blocks impregnated with the highest concentrations of preservative.

Similarly, the mean loss in mass of untreated heartwood control blocks, after exposure to both strains of *N. lepideus* (tables 2.3.2.3 and 2.3.2.5), was reduced compared with respective untreated *virulence controls* (table 2.3.2.1). However, this reduction was less pronounced than for sapwood blocks and was not associated with higher preservative concentrations in adjacent treated heartwood blocks. The mean loss in mass of heartwood *virulence control* blocks (table 2.3.2.1) and control blocks after exposure to ***C. puteana*** (table 2.3.2.7) showed that this basidiomycete was again largely unaffected by any movement of preservative from adjacent treated wood blocks at retentions of less than 6.78 kg/m^3.

2.3.2.4. Toxicity of treated wood to Moulds.

The growth of ***C. resinae*** and ***T. polysporum*** on treated sapwood blocks was prevented by concentrations of applied preservative substantially higher than those required to prevent basidiomycete decay of treated sapwood blocks (table 2.3.2.11). The former mould displayed a greater resistance to the preservative.

Comparison of the mean scores for the growth of ***T. polysporum*** on *virulence control* blocks (table 2.3.2.1) and on control blocks in the same culture vessel as preservative treated blocks (table 2.3.2.10) indicates that the virulence of this mould, as for the majority of basidiomycetes (section 2.3.2.3.), was reduced by a possible transfer of preservative from the treated blocks at higher preservative concentrations.

2.3.3. Distribution of fluoride and chromium in treated poles.

2.3.3.1. Introduction.

Wood samples were excised from 8 positions within the uncreosoted area on 2 discs cut from the treated zone of each of 10 pole sections after exposure at a field site for up to 20 months after Rentex treatment (section 2.2.2.2.2. and figure 2.2.1). The samples were analysed for fluoride and chromium content (section 2.2.2.3.) to establish the distribution of these preservative elements in remedially treated timber.

2.3.3.2. Tables and figures of analysis measurements.

Concentrations of fluoride and chromium in wood samples, 1-8 (figure 2.2.1), from each sample disc recovered from pole sections 1 and 2; 3 and 4; 5 and 6; 7 and 8; and 9 and 10; (additional data for which are shown in table 2.3.3.1) are detailed in tables 2.3.3.2, 2.3.3.3, 2.3.3.4, 2.3.3.5 and 2.3.3.6 respectively. For purposes of comparison these fluoride and chromium data (tables 2.3.3.2 - 2.3.3.6) were combined for wood samples from similar sample disc heights from pole sections subjected to identical periods of field exposure (figures 2.3.3.1, 2.3.3.2 and 2.3.3.3).

The mean concentrations of fluoride and chromium for wood samples, 1-8, for each sample disc height (figures 2.3.3.1, 2.3.3.2 and 2.3.3.3) were combined to produce a mean concentration for all wood samples from each disc height for each period of field exposure in figure 2.3.3.4, parts A and B. The concentrations of fluoride and chromium, displayed for each disc height in figure 2.3.3.4, parts A and B, were then combined to display the total mean concentration of each preservative element in treated timber, after each period of field exposure, in figure 2.3.3.4, part C.

An analysis of variance was carried out using the fluoride concentrations of wood samples from positions 2, 3, 5, 6, 7 and 8 (tables 2.3.3.2 - 2.3.3.6), ie. excluding samples 1 and 4 which included the preservative injection site (figure 2.2.1). Wood sample position and duration of field exposure were identified as significant factors influencing fluoride distribution in the treated timber. The mean fluoride concentrations for these factors, fitted by the statistical analysis, are presented in table 2.3.3.7. A significant interaction of the factors was also identified and table 2.3.3.8 displays the fitted mean fluoride concentrations for this interaction.

2.3.3.3. Mean fluoride and chromium concentrations at injection sites.

As the bulk of the applied fluoride and chromium clearly remained at the sites of preservative injection, positions 1 and 4 (tables 2.3.3.2 - 2.3.3.6 and figures 2.3.3.1 - 2.3.3.3), concentrations of each element at these sample positions (figures 2.3.3.1 - 2.3.3.3) were combined to produce a separate mean concentration of fluoride and chromium for the entire injection site of each groundline position at each period after treatment and for both groundline positions combined at each period after treatment (data not shown). Statistical comparisons of these mean values were carried out using oneway analysis of variance. Where oneway statistical comparisons of more than 2 values indicated significant differences, Scheffes S test for analysis of contrasts (Dowdy and Wearden, 1991) was employed to identify where these existed.

At 0 (no field exposure) and 2 months after remedial treatment, the mean fluoride concentrations for injection sites above the groundline were significantly greater than concentrations of this element for injection sites below the groundline, $P = 0.008$ and 0.042 respectively (figure 2.3.3.1). For the same comparison, the mean chromium concentration above the groundline was significantly greater than that concentration below the groundline at 0 months after treatment, $P = 0.008$ (figure 2.3.3.1). At no time after treatment were

Table 2.3.3.1. Period of field exposure, diameter, no. of injections and percentage moisture content 175 mm above and below the groundline of pole sections 1-10.

Pole Number	Period of Field Exposure (months)	Moisture content (%) of sample disc at 175 mm above	below	Diameter (cm)	Number of Preservative Injections
1	0	21.0	21.0	21.2	91
2	0	14.0	14.0	18.8	84
3	2	16.0	18.0	18.5	70
4	2	17.0	17.0	19.7	77
5	5	22.0	24.0	22.6	91
6	5	29.0	30.0	16.2	63
7	12	26.5	28.0	23.6	98
8	12	16.0	18.0	19.4	84
9	20	24.0	22.0	20.8	84
10	20	21.0	23.0	21.6	98

Table 2.3.3.2. Mean percentage fluoride and chromium concentrations (w/w) of wood samples 1-8 at 0 months (1 week) after remedial treatment (standard deviations for means of 2 in parenthesis).

Pole Section Number	Disc Position Relative to Groundline	Sample Position	Mean Percentage Fluoride Concentration	Mean Percentage Chromium Concentration
1	175 mm above	1	2.9086 (0.0180)	2.3760 (0.0218)
		2	0.1041 (0.0094)	0.0513 (0.0038)
		3	0.1244 (0.0013)	0.0669 (0.0102)
		4	2.0530 (0.0038)	1.9660 (0.1620)
		5	0.0430 (0.0924)	0.0072 (0.0067)
		6	0.0285 (0.0001)	0.0021 (0.0002)
		7	0.0391 (0.0001)	0.0032 (0.0006)
		8	0.0552 (0.0000)	0.0060 (0.0037)
	175 mm below	1	1.4155 (0.0687)	1.2440 (0.1189)
		2	0.0286 (0.0000)	0.0001 (0.0001)
		3	0.0295 (0.0021)	0.0004 (0.0001)
		4	0.5596 (0.0011)	0.5403 (0.0020)
		5	0.0289 (0.0022)	0.0000 (0.0000)
		6	0.0297 (0.0004)	0.0024 (0.0034)
		7	0.0272 (0.0001)	0.0006 (0.0008)
		8	0.0272 (0.0004)	0.0000 (0.0000)
2	175 mm above	1	1.8350 (0.1560)	1.8289 (0.0799)
		2	0.0456 (0.0108)	0.0042 (0.0009)
		3	0.0388 (0.0002)	0.0019 (0.0009)
		4	0.4677 (0.0004)	0.2498 (0.0253)
		5	0.0294 (0.0002)	0.0042 (0.0009)
		6	0.0302 (0.0007)	0.0045 (0.0014)
		7	0.0331 (0.0004)	0.0013 (0.0018)
		8	0.0251 (0.0000)	0.0038 (0.0000)
	175 mm below	1	0.5282 (0.0192)	0.2790 (0.0622)
		2	0.0888 (0.0000)	0.0104 (0.0000)
		3	0.0338 (0.0002)	0.0038 (0.0004)
		4	0.1817 (0.0002)	0.0478 (0.0009)
		5	0.0474 (0.0042)	0.0004 (0.0006)
		6	0.0741 (0.0000)	0.0287 (0.0000)
		7	0.0329 (0.0004)	0.0005 (0.0008)
		8	0.0227 (0.0000)	0.0001 (0.0000)

Table 2.3.3.3. Mean percentage fluoride and chromium concentrations (w/w) of wood samples 1-8 at 2 months after remedial treatment (standard deviations for means of 2 in parenthesis).

Pole Section Number	Disc Position Relative to Groundline	Sample Position	Mean Percentage Fluoride Concentration	Mean Percentage Chromium Concentration
3	175 mm above	1	1.6035 (0.1286)	0.3924 (0.0302)
		2	0.0444 (0.0001)	0.0000 (0.0000)
		3	0.0324 (0.0001)	0.0000 (0.0000)
		4	1.7022 (0.0005)	0.4297 (0.0254)
		5	0.0295 (0.0000)	0.0000 (0.0000)
		6	0.0328 (0.0008)	0.0000 (0.0000)
		7	0.2075 (0.0009)	0.0331 (0.0019)
		8	0.0357 (0.0051)	0.0006 (0.0008)
	175 mm below	1	1.3209 (0.1316)	0.5430 (0.0091)
		2	0.0641 (0.0000)	0.0008 (0.0003)
		3	0.0475 (0.0002)	0.0001 (0.0000)
		4	1.4252 (0.0071)	0.5880 (0.0128)
		5	0.2273 (0.0006)	0.0049 (0.0022)
		6	0.0376 (0.0001)	0.0001 (0.0001)
		7	0.0637 (0.0014)	0.0012 (0.0004)
		8	0.0486 (0.0001)	0.0005 (0.0007)
4	175 mm above	1	2.2075 (0.0420)	0.9220 (0.1570)
		2	0.0623 (0.0001)	0.0001 (0.0001)
		3	0.0425 (0.0001)	0.0000 (0.0000)
		4	0.7787 (0.0009)	0.2410 (0.0200)
		5	0.1178 (0.0001)	0.0002 (0.0001)
		6	0.0445 (0.0001)	0.0004 (0.0001)
		7	0.0532 (0.0005)	0.0002 (0.0003)
		8	0.0393 (0.0005)	0.0000 (0.0000)
	175 mm below	1	0.9270 (0.0290)	0.4791 (0.0538)
		2	0.0417 (0.0027)	0.0000 (0.0000)
		3	0.0401 (0.0001)	0.0000 (0.0000)
		4	0.3865 (0.0139)	0.0940 (0.0145)
		5	0.0429 (0.0026)	0.0002 (0.0003)
		6	0.0310 (0.0006)	0.0002 (0.0003)
		7	0.0287 (0.0000)	0.0000 (0.0000)
		8	0.0319 (0.0001)	0.0003 (0.0003)

Table 2.3.3.4. Mean percentage fluoride and chromium concentrations (w/w) of wood samples 1-8 at 5 months after remedial treatment (standard deviations for means of 2 in parenthesis).

Pole Section Number	Disc Position Relative to Groundline	Sample Position	Mean Percentage Fluoride Concentration	Mean Percentage Chromium Concentration
5	175 mm above	1	0.3576 (0.0366)	0.1688 (0.0163)
		2	0.0754 (0.0010)	0.0021 (0.0004)
		3	0.0283 (0.0004)	0.0001 (0.0001)
		4	0.0652 (0.0007)	0.0014 (0.0002)
		5	0.0312 (0.0005)	0.0002 (0.0003)
		6	0.0020 (0.0003)	0.0008 (0.0012)
		7	0.0128 (0.0001)	0.0007 (0.0008)
		8	0.0114 (0.0000)	0.0003 (0.0001)
	175 mm below	1	0.5636 (0.0016)	0.1424 (0.0323)
		2	0.0896 (0.0023)	0.0013 (0.0000)
		3	0.0161 (0.0017)	0.0000 (0.0001)
		4	0.0659 (0.0006)	0.0008 (0.0002)
		5	0.0193 (0.0037)	0.0004 (0.0002)
		6	0.0209 (0.0001)	0.0003 (0.0004)
		7	0.0149 (0.0001)	0.0002 (0.0002)
		8	0.0024 (0.0000)	0.0002 (0.0002)
6	175 mm above	1	0.9308 (0.0118)	0.3718 (0.0113)
		2	0.0490 (0.0017)	0.0005 (0.0001)
		3	0.0306 (0.0021)	0.0003 (0.0000)
		4	1.5915 (0.0712)	0.5696 (0.0483)
		5	0.1606 (0.0000)	0.0023 (0.0000)
		6	0.0315 (0.0038)	0.0005 (0.0002)
		7	0.4396 (0.0119)	0.3174 (0.0067)
		8	0.4448 (0.0000)	0.2805 (0.0000)
	175 mm below	1	1.0102 (0.0013)	0.5035 (0.0066)
		2	0.0631 (0.0007)	0.0008 (0.0002)
		3	0.0192 (0.0005)	0.0002 (0.0001)
		4	0.7385 (0.0296)	0.5387 (0.0035)
		5	0.1047 (0.0032)	0.0032 (0.0004)
		6	0.0476 (0.0015)	0.0006 (0.0001)
		7	0.2268 (0.0360)	0.0395 (0.0022)
		8	0.2253 (0.0166)	0.0367 (0.0016)

Table 2.3.3.5. Mean percentage fluoride and chromium concentrations (w/w) of wood samples 1-8 at 12 months after remedial treatment (standard deviations for means of 2 in parenthesis).

Pole Section Number	Disc Position Relative to Groundline	Sample Position	Mean Percentage Fluoride Concentration	Mean Percentage Chromium Concentration
7	175 mm above	1	0.4745 (0.0657)	0.1841 (0.0082)
		2	0.0816 (0.0020)	0.0005 (0.0000)
		3	0.0685 (0.0009)	0.0003 (0.0003)
		4	0.3088 (0.0037)	0.0238 (0.0062)
		5	0.0792 (0.0005)	0.0003 (0.0000)
		6	0.0457 (0.0009)	0.0000 (0.0000)
		7	0.0330 (0.0024)	0.0001 (0.0002)
		8	0.0203 (0.0000)	0.0001 (0.0000)
	175 mm below	1	0.6818 (0.0773)	0.2864 (0.0375)
		2	0.1505 (0.0016)	0.0027 (0.0005)
		3	0.1051 (0.0015)	0.0005 (0.0001)
		4	0.3698 (0.0035)	0.1460 (0.0210)
		5	0.1321 (0.0005)	0.0011 (0.0000)
		6	0.0912 (0.0040)	0.0005 (0.0004)
		7	0.0444 (0.0016)	0.0004 (0.0006)
		8	0.0174 (0.0003)	0.0003 (0.0003)
8	175 mm above	1	1.2260 (0.2850)	0.7210 (0.1790)
		2	0.0926 (0.0040)	0.0012 (0.0006)
		3	0.0502 (0.0037)	0.0005 (0.0001)
		4	0.6998 (0.0138)	0.3615 (0.0427)
		5	0.0526 (0.0018)	0.0000 (0.0000)
		6	0.0571 (0.0015)	0.0003 (0.0003)
		7	0.2063 (0.0319)	0.0269 (0.0008)
		8	0.2418 (0.0962)	0.0369 (0.0198)
	175 mm below	1	0.8082 (0.1004)	0.2698 (0.0675)
		2	0.0648 (0.0035)	0.0004 (0.0003)
		3	0.0330 (0.0001)	0.0001 (0.0001)
		4	0.3865 (0.0167)	0.2247 (0.0019)
		5	0.0843 (0.0041)	0.0006 (0.0000)
		6	0.0454 (0.0031)	0.0005 (0.0004)
		7	0.0956 (0.0065)	0.0026 (0.0003)
		8	0.0209 (0.0004)	0.0000 (0.0000)

Table 2.3.3.6. Mean percentage fluoride and chromium concentrations (w/w) of wood samples 1-8 at 20 months after remedial treatment (standard deviations for means of 2 in parenthesis).

Pole Section Number	Disc Position Relative to Groundline	Sample Position	Mean Percentage Fluoride Concentration	Mean Percentage Chromium Concentration
9	175 mm above	1	1.4111 (0.0398)	0.7920 (0.0223)
		2	0.0672 (0.0042)	0.0015 (0.0001)
		3	0.0605 (0.0067)	0.0018 (0.0001)
		4	0.6828 (0.0619)	0.3059 (0.0024)
		5	0.0393 (0.0006)	0.0012 (0.0001)
		6	0.0126 (0.0006)	0.0006 (0.0001)
		7	0.0235 (0.0019)	0.0002 (0.0000)
		8	0.0093 (0.0001)	0.0008 (0.0001)
	175 mm below	1	1.3486 (0.0383)	0.7524 (0.0038)
		2	0.0573 (0.0009)	0.0006 (0.0001)
		3	0.0230 (0.0015)	0.0009 (0.0001)
		4	1.4689 (0.0662)	0.7895 (0.0011)
		5	0.0698 (0.0015)	0.0009 (0.0001)
		6	0.0384 (0.0055)	0.0006 (0.0001)
		7	0.1778 (0.0102)	0.1038 (0.0031)
		8	0.0319 (0.0015)	0.0118 (0.0005)
10	175 mm above	1	0.0324 (0.0009)	0.0201 (0.0011)
		2	0.0134 (0.0010)	0.0006 (0.0000)
		3	0.0150 (0.0017)	0.0006 (0.0001)
		4	0.1012 (0.0059)	0.0402 (0.0006)
		5	0.0101 (0.0019)	0.0005 (0.0000)
		6	0.0127 (0.0007)	0.0007 (0.0001)
		7	0.0124 (0.0000)	0.0009 (0 .0003)
		8	0.0140 (0.0001)	0.0006 (0.0001)
	175 mm below	1	0.0605 (0.0057)	0.0268 (0.0005)
		2	0.0177 (0.0012)	0.0016 (0.0001)
		3	0.0124 (0.0014)	0.0011 (0.0001)
		4	0.0531 (0.0041)	0.0293 (0.0006)
		5	0.0138 (0.0002)	0.0255 (0.0000)
		6	0.0232 (0.0027)	0.0016 (0.0001)
		7	0.0139 (0.0003)	0.0010 (0.0001)
		8	0.0137 (0.0013)	0.0010 (0.0000)

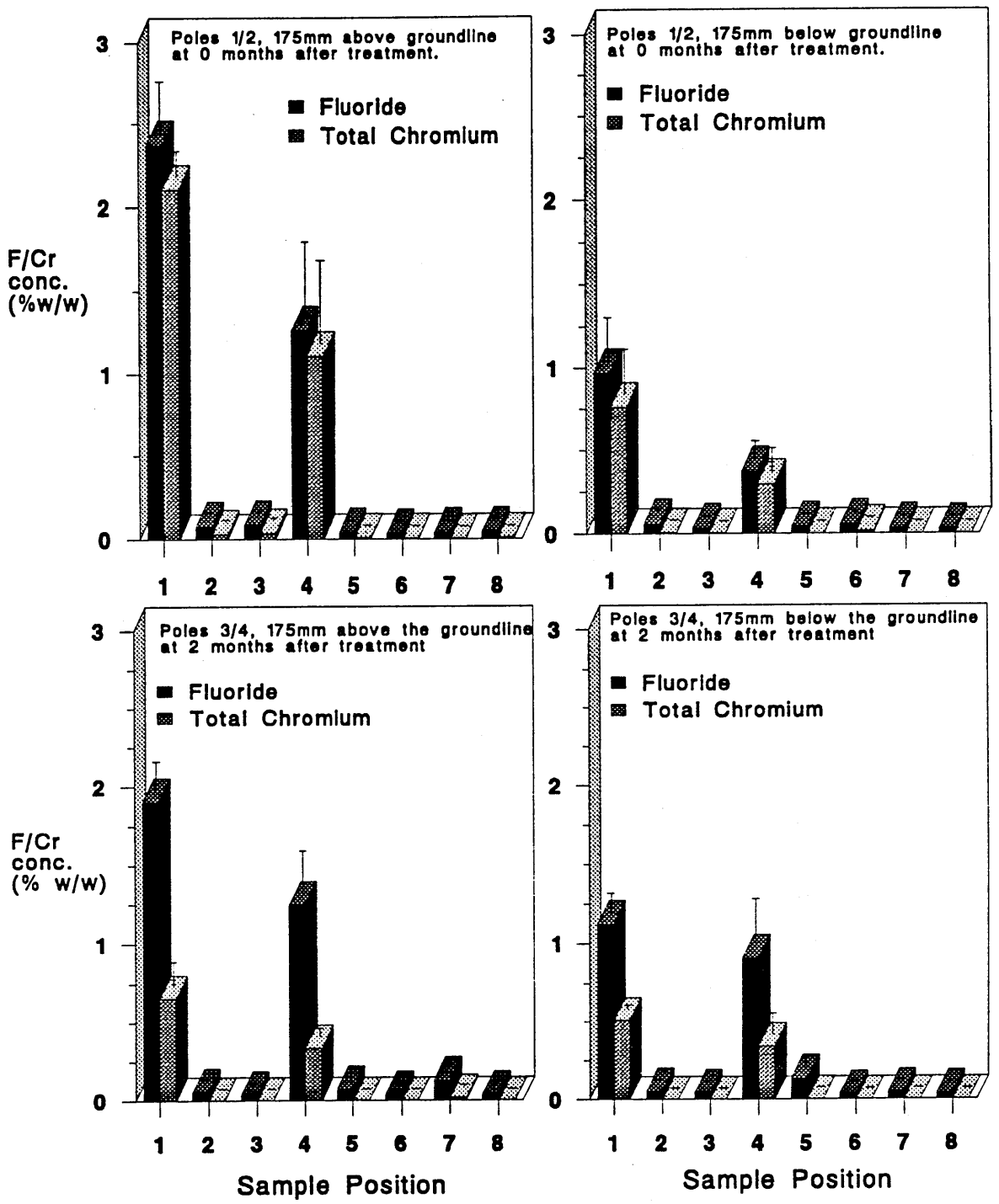


Figure 2.3.3.1. Mean fluoride and chromium concentrations (% w/w) of each sample position 1 - 8, combined for each sample disc height, 175 mm above and below the groundline, of pole sections 1 and 2 immediately after remedial treatment (0 months), and pole sections 3 and 4 at 2 months after remedial treatment. Standard error bars are for means of 4.

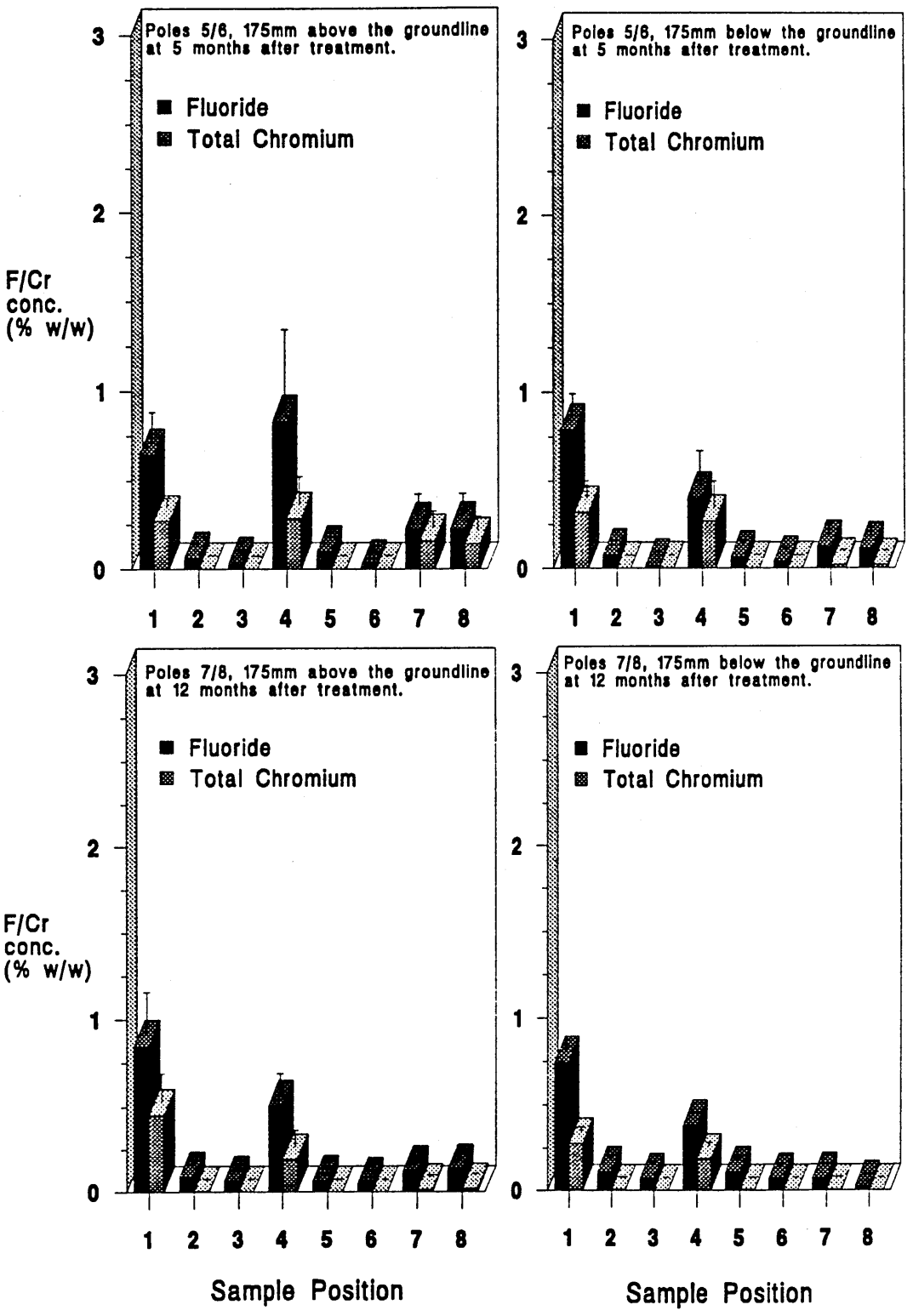


Figure 2.3.3.2. Mean fluoride and chromium concentrations (% w/w) of each sample position 1 - 8, combined for each sample disc height, 175 mm above and below the groundline, of pole sections 5 and 6 at 5 months after remedial treatment, and pole sections 7 and 8 at 12 months after remedial treatment. Standard error bars are for means of 4.

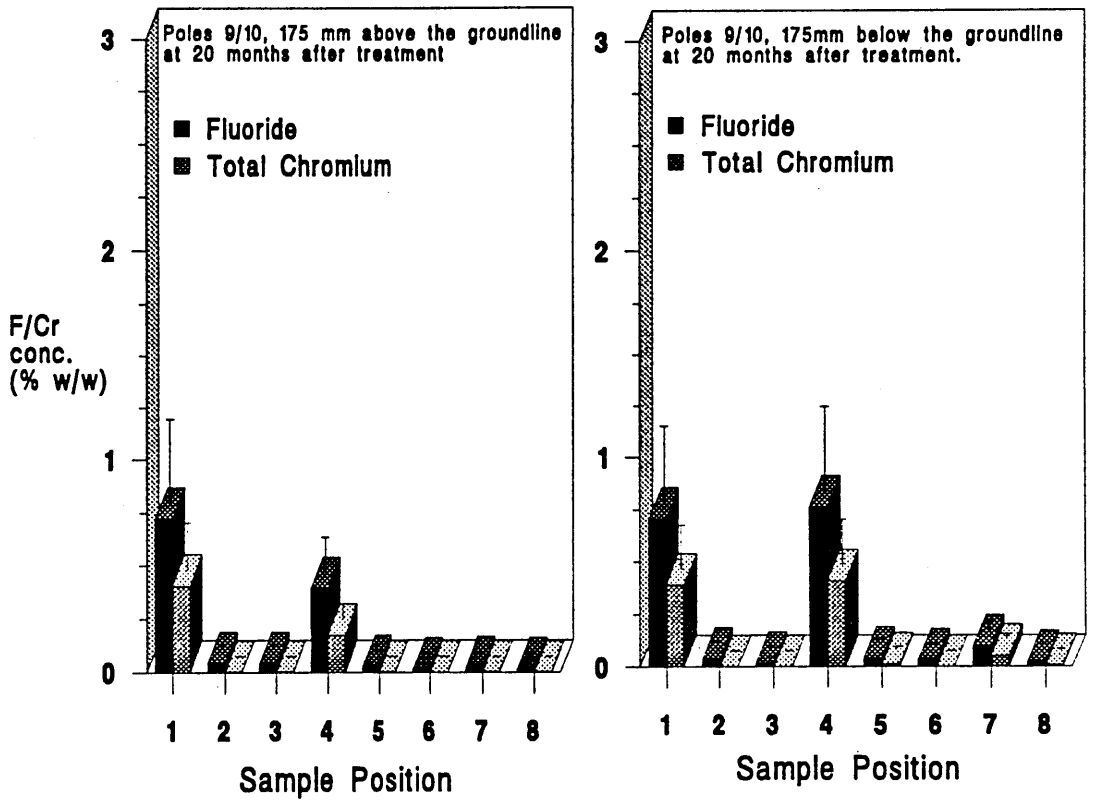


Figure 2.3.3.3. Mean fluoride and chromium concentrations (% w/w) of each sample position 1 - 8, combined for each sample disc height, 175 mm above and below the groundline, of pole sections 9 and 10 at 20 months after remedial treatment. Standard error bars are for means of 4.

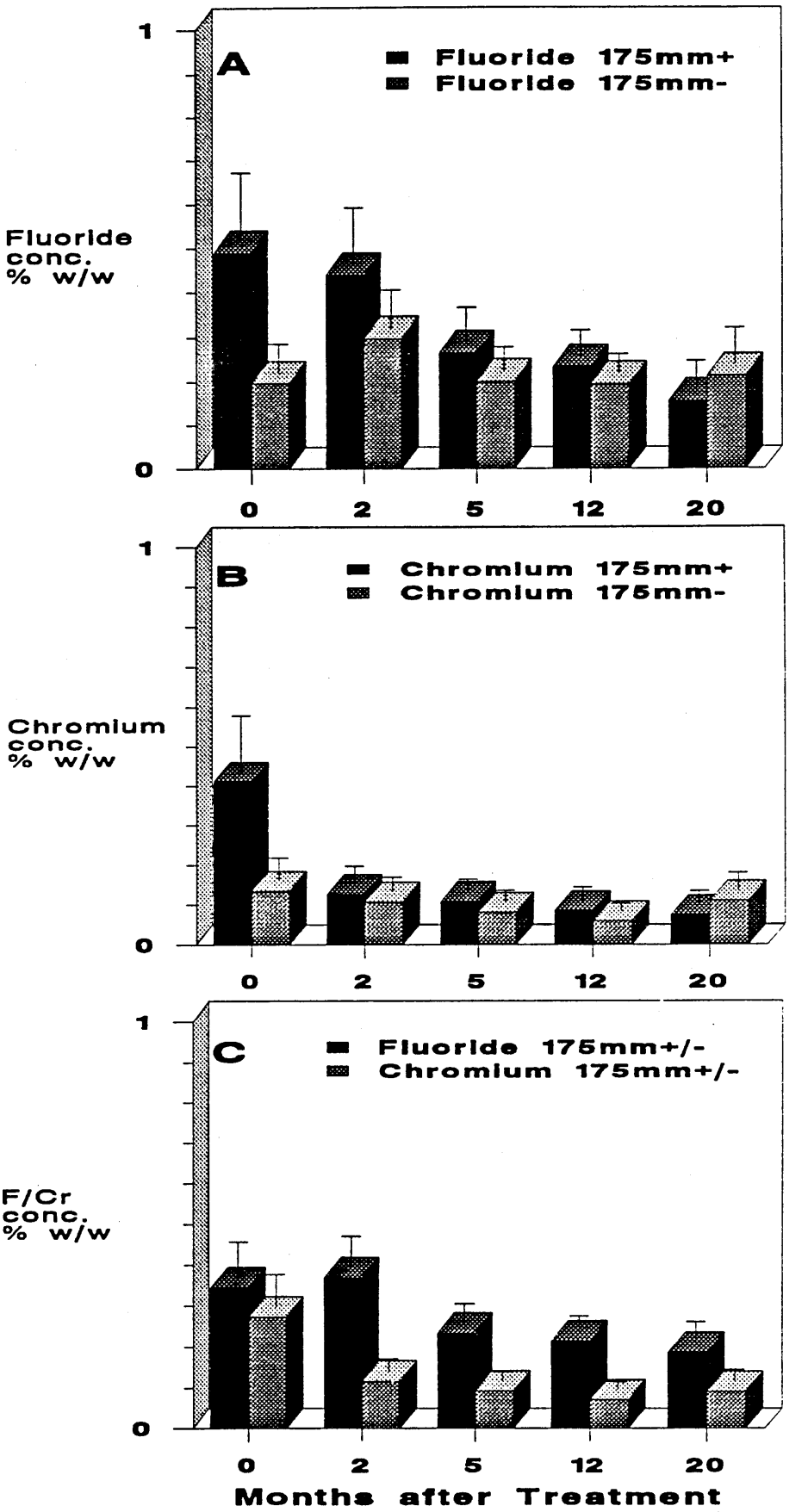


Figure 2.3.3.4. A, B and C. Mean fluoride and chromium concentrations (% w/w), combined for all sample positions 1 - 8, of each sample disc, 175 mm above (+) and below (-) the groundline, and for both sample discs (+/-), of pole sections 1/2, 3/4, 5/6, 7/8 and 9/10 at 0, 2, 5, 12 and 20 months after remedial treatment respectively. Standard error bars are for means of 32 (A, B) and 64 (C).

Table 2.3.3.7. Fitted mean percentage fluoride concentrations (w/w) and 95 % confidence intervals for the main effects of field exposure duration and sample position.

Duration of Field Exposure (months)	Fitted Mean % Fluoride Concentration	95 % Confidence Intervals
0	0.0463	(0.0401 - 0.0529)
2	0.0509	(0.0439 - 0.0584)
5	0.0216	(0.0167 - 0.0272)
12	0.0545	(0.0479 - 0.0616)
20	0.0214	(0.0174 - 0.0257)

Sample Position	Fitted Mean % Fluoride Concentration	95 % Confidence Intervals
2	0.0587	(0.0517 - 0.0662)
3	0.0384	(0.0329 - 0.0444)
5	0.0489	(0.0423 - 0.0560)
6	0.0334	(0.0283 - 0.0389)
7	0.0307	(0.0249 - 0.0370)
8	0.0206	(0.0162 - 0.0256)

Table 2.3.3.8. Fitted mean percentage fluoride concentrations (w/w) and 95% confidence intervals for the interaction of sample position and duration of field exposure.

Duration of Field Exposure (months)	Sample Position	Fitted Mean % Fluoride Concentration	95 % Confidence Intervals
0	2	0.0685	(0.0522 - 0.0870)
	3	0.0564	(0.0417 - 0.0733)
	5	0.0411	(0.0411 - 0.0556)
	6	0.0433	(0.0306 - 0.0582)
	7	0.0372	(0.0254 - 0.0511)
	8	0.0355	(0.0241 - 0.0491)
2	2	0.0587	(0.0436 - 0.0761)
	3	0.0458	(0.0326 - 0.0612)
	5	0.0649	(0.0468 - 0.0860)
	6	0.0414	(0.0289 - 0.0561)
	7	0.0526	(0.0365 - 0.0716)
	8	0.0438	(0.0310 - 0.0590)
5	2	0.0590	(0.0435 - 0.0769)
	3	0.0178	(0.0098 - 0.0282)
	5	0.0389	(0.0252 - 0.0557)
	6	0.0163	(0.0087 - 0.0262)
	7	0.0121	(0.0041 - 0.0244)
	8	0.0050	(0.0006 - 0.0136)
12	2	0.0804	(0.0605 - 0.1032)
	3	0.0594	(0.0444 - 0.0761)
	5	0.0821	(0.0645 - 0.1019)
	6	0.0563	(0.0419 - 0.0729)
	7	0.0493	(0.0339 - 0.0678)
	8	0.0164	(0.0081 - 0.0276)
20	2	0.0327	(0.0219 - 0.0456)
	3	0.0231	(0.0142 - 0.0341)
	5	0.0269	(0.0173 - 0.0388)
	6	0.0188	(0.0109 - 0.0289)
	7	0.0146	(0.0069 - 0.0252)
	8	0.0148	(0.0079 - 0.0238)

mean concentrations of either element at injection sites below the groundline significantly greater than concentrations above the groundline.

Above the groundline, mean concentrations of fluoride at injection sites, at 0 and 2 months after remedial treatment, were not significantly different (figure 2.3.3.1), but both were significantly greater than those fluoride concentrations above the groundline at 5, 12 and 20 months, $P = 0.001$ (figures 2.3.3.2 and 2.3.3.3). Similarly the mean chromium concentration above the groundline at 0 months after remedial treatment was significantly greater than at 2, 5, 12 and 20 months, $P < 0.0005$ (figures 2.3.2.1 - 2.2.3.3). In contrast, at injection sites below the groundline, there were no significant differences between periods of field exposure for mean concentrations of either element.

Mean concentrations of fluoride, combined for injection sites above and below the groundline, were significantly greater at 0 and 2 months after treatment than those at 5, 12 and 20 months, $P = 0.002$. For the same comparison, the mean concentration of chromium at 0 months after treatment was significantly greater than those at 2, 5, 12 and 20 months, $P < 0.0005$.

The significantly higher mean concentrations of fluoride, at 0 and 2 months, and chromium, at 0 months, for injection sites above the groundline compared with injection sites below the groundline, may have been caused by greater leverage allowed for the action of the mechanical pump when treating this area. This may have resulted in greater applied pressure and needle penetration beyond the creosoted band and therefore greater deposition of preservative. The significant early reduction in the concentrations of both elements, particularly chromium, for injection sites above the groundline, coupled with the stability of element concentrations below the groundline, over the course of the experiment, served to remove this possible treatment difference and ensured a significant reduction in mean concentrations of each element, as time after treatment increased.

2.3.3.4. Mean fluoride and chromium concentrations at all sample positions.

A series of statistical comparisons, as detailed briefly in section 2.3.3.3., was carried out to identify any significant differences for mean concentrations of fluoride or chromium, combined for all sample positions 1 - 8 (figure 2.3.3.4, parts A, B and C), between groundline positions and periods of field exposure after treatment.

There were no significant differences between mean concentrations of fluoride above and below the groundline at any time after remedial treatment (figure 2.3.3.4, part A). This was also the case for comparisons of these chromium concentrations (figure 2.3.3.4, part B).

Similarly, there were no significant differences in the mean concentrations of fluoride, above or below the groundline, between periods of field exposure (figure 2.3.3.4, part A). Though mean concentrations of chromium below the groundline were, like fluoride, not significantly different between periods of field exposure, the mean concentration of chromium above the groundline, at 0 months was significantly and progressively greater than that at 5, 12 and 20 months after remedial treatment, $P = 0.043$, 0.031 and 0.026 respectively (figure 2.3.3.4, part B).

The mean fluoride concentration, combined for both groundline positions at 2 months after treatment was significantly greater than that at 20 months after treatment, $P = 0.048$ (figure 2.3.3.4, part C), while the mean concentration of chromium, combined for both groundline positions, was significantly greater at 0 months than that at 5, 12 and 20 months, $P = 0.030$, 0.014 and 0.021 respectively (figure 2.3.3.4, part C).

Mean concentrations of both elements below the groundline in these pole sections therefore remained relatively stable throughout the period of the experiment (figure 2.3.3.4, parts A and B). This contrasts with the findings for fluoride and chromium concentration

above the groundline, which displayed a pronounced reduction, 5 and 2 months after treatment respectively, which continued gradually, becoming significant for chromium 5, 12 and 20 months after treatment (figure 2.3.3.4, parts A and B). These significant reductions in chromium concentrations above the groundline (figure 2.3.3.4, part B) ensured that significant reductions also occurred, for concentrations of this element, combined for both groundline positions, at 5, 12 and 20 months after treatment (figure 2.3.3.4, part C). However, the significant reduction in the mean concentration of fluoride, for both groundline positions combined, between 2 and 20 months after treatment (figure 2.3.3.4, part C) was due to a measurable though not significant increase below the groundline and a slight non-significant decrease above the groundline at 2 months after treatment (figure 2.3.3.4, part A).

Mean concentrations of chromium in treated pole sections therefore suffered a severe reduction, after 2 months field exposure, which was maintained over the course of the experiment. Fluoride concentrations displayed more gradual reductions with increasing time after treatment. Figure 2.3.3.4, part A, indicates that a movement of fluoride from above the groundline of the remedially treated area served to maintain the concentrations of this element below the groundline.

2.3.3.5. Fitted mean fluoride concentrations for sample positions distant from injection sites using a statistical model.

Tables 2.3.3.2 - 2.3.3.6 and figures 2.3.3.1 - 2.3.3.3 indicate that a proportion of fluoride, though not chromium, appeared to diffuse from the injection sites, positions 1 and 4, to sample positions 2, 3, 5, 6, 7 and 8 (figure 2.2.1). The higher mean concentrations of total chromium away from the injection site at sample positions 7 and 8 above the groundline for the combined chromium values of pole sections 5 and 6 (figure 2.3.3.2), were most likely due to the small diameter of pole section 6 (table 2.3.3.1.), allowing

greater penetration of the injection needle and extending the injection line to these sample positions.

Therefore, using the fluoride values at positions 2, 3, 5, 6, 7 and 8, an analysis of variance was undertaken to construct a statistical model to determine the influence of the factors of sample position, time after treatment and sample disc height on the diffusion of fluoride from the injection site. The influence of the co-variates of moisture content, pole diameter and number of preservative injections applied to each pole were also examined by their inclusion in the analysis.

Throughout the analysis, 3 statistical checks were made to establish the validity of the findings. Firstly, the standardised residual values generated by the statistical model were examined. A standardised residual value, the measure of the goodness of fit of a fitted to an actual value, is defined as the actual value (y) minus the statistically fitted value (y^) divided by the standard deviation of y - y^ (Draper and Smith, 1981). For appropriate models, standardised residual values should follow a standard normal distribution with 95 % of values lying within the range of -2 to 2 and 99.8 % within the range of -3 to 3. Therefore, if more than 5 % of the observations had standardised residual values outside the range of -2 to 2 the model findings were invalidated. Secondly, a plot of residual values against fitted values was carried out to confirm the lack of systematic trends by an even horizontal spread about 0 and finally, a histogram of residual values was constructed to validate the normal distribution assumption.

Initial statistical analysis indicated that groundline sample disc height and the co-variates of pole diameter and number of injections had no significant bearing on the mean levels of diffused fluoride found. In the absence of these variables, duration of field exposure, sample position, their interaction and the co-variate of moisture content were found to have a significant influence, P = 0.001, 0.003, 0.005 and 0.013 respectively. However, the number and magnitude of unusual standard residual values proved unacceptably large. The plot of

residual values as a function of fitted values indicated 22 outliers above a fluoride concentration of 0.14 %, representing 9 % of the total fluoride values in the analysis. These values were subsequently removed from the analysis as the higher than expected chromium levels at these sample positions could not be explained by chromium diffusion, and this indicated that fluoride deposition had most likely also occurred by means other than diffusion. An improved significance for the same factors and co-variate was indicated. Though the number of unusual observations represented only 5.5 % of the total, 12 out of 218, 2 of the standard residual values were above 3, representing 0.9 % of the total, whereas 0.2 % would represent an acceptable level. The frequency histogram of residual values was skewed to the left confirming the departure of these values from a normal distribution. In order to normalise the data, a square root transformation was carried out. The factors of exposure period, sample position and their interaction were very highly significant at $P < 0.0005$, 0.0005 and $P = 0.001$ respectively. The co-variate moisture content was significant at $P = 0.001$, an increase in moisture content was indicative of increased diffused fluoride concentrations.

The validity of the findings was confirmed as unusual observations were within an acceptable level, the residual versus fitted value plot displayed an even horizontally spread distribution and the histogram indicated that residual values were normally distributed.

Table 2.3.3.7. shows the fitted mean fluoride concentrations for each time (duration of field exposure) and sample position. The mean percentage fluoride concentration for all sample positions combined at 0, 2 and 12 months after treatment was significantly greater than at 5 and 20 months. Thus, an increasing mean fluoride value up to 12 months after preservative treatment prior to a reduction after 20 months was interrupted by a low mean value at 5 months after treatment (table 2.3.3.7). The low value at 5 months came about for 2 reasons. Firstly, the removal from the statistical analysis of 10 fluoride values above 0.14% for pole section 6 (table 2.3.3.4). These values were evidently due to deposition by the injection process, as needle penetration in this pole section was increased as a

consequence of its smaller diameter (table 2.3.3.1.). Secondly, mean fluoride and chromium concentrations for sample position 4 of pole section 5 at the same exposure period (table 2.3.3.4.) indicated that injected preservative had not reached this position in any great quantity and diffusion of fluoride to positions 5, 6, 7 and 8 was consequently reduced. As the mean value of fluoride at 5 months (table 2.3.3.7.) was essentially based on these reduced values its representativeness is therefore doubtful.

The mean percentage fluoride concentration at sample position 2, combined over all periods of field exposure, was significantly greater than at positions 3, 6, 7 and 8 (table 2.3.3.7). Similarly, the fluoride value for sample position 5 was significantly greater than for positions 6, 7 and 8, whilst the mean values for positions 3 and 6 were significantly greater than for position 8. This pattern of fluoride deposition strongly indicated transverse movement primarily through the tracheids according to a concentration gradient from injection site sample positions 1 and 4 (figure 2.2.1, section 2.2.2.2.2.) to positions 2 and 5 and thereafter to 3 and 6. By virtue of their orientation to, and distance from, the injection site, the possible reliance of sample positions 7 and 8 on 'secondary' radial diffusion of fluoride through ray/parenchyma cells from sample positions 5 and 6 resulted in the lower mean values of fluoride at the pole centre.

Fitted mean percentage fluoride concentrations of sample positions 2, 3, 5, 6, 7 and 8 after each period of field exposure are presented in table 2.3.3.8. The significant interaction of these 2 factors in their effect on fluoride concentrations, $P = 0.001$, indicated that neither factor operated independently of the other. Table 2.3.3.8. indicates that significant differences between different sample positions after the same period of exposure and between the same sample positions after different exposure periods were respectively as follows:

Period of Exposure	Sample Position
0	2 > 7, 8
2	No significant differences
5	2 > 3, 6, 7, 8; 5 > 7, 8
12	2, 3, 5, 6, 7 > 8
20	No significant differences

Sample Position	Period of Exposure
2	12 > 20
3	0, 2, 12 > 5; 0, 12 > 20
5	0, 2, 12 > 20; 12 > 0, 5
6	0, 2, 12 > 5; 0, 12 > 20
7	0, 2, 12 > 5, 20
8	0, 2 > 5, 20; 2 > 12

Where significant differences existed for this interaction, the trends found agreed with those for the main effects of each factor. For instance, significant differences between sample positions at each period of exposure indicated that sample positions 7 and 8 retained lower mean concentrations of fluoride. Similarly, significant differences in the mean concentrations of fluoride at each sample position between exposure periods confirmed the higher and lower concentrations of fluoride at 12 and 20 months after treatment respectively. Again the findings for 5 months after remedial treatment were questionable for the reasons stated earlier and more so as the mean fluoride values for the interaction were based on a greatly reduced number of observations.

Mean percentage fluoride concentrations at 0 months (table 2.3.3.8) appear to be inconsistent with the main effects of each factor (table 2.3.3.7), with concentrations of fluoride at position 3 > position 5, and position 5 < than position 6. However, sample positions 2 and 3 above the groundline of pole section 1 and sample position 6 below the

groundline of pole section 2 at this time (table 2.3.3.2) contained relative concentrations of fluoride and chromium indicative of preservative injection rather than fluoride diffusion alone. This indicated preservative entry to these positions via cracks or splits adjacent to the injection site. These higher than usual fluoride levels were not in excess of 0.14 % and were not removed from the statistical analysis. These few unusual values would have a pronounced effect on mean fluoride concentrations for the interaction of time and sample position (table 2.3.3.8), based on up to 8 observations only. However, the mean fluoride concentrations for the main effects of exposure period and sample position (table 2.3.3.7) were based on up to approximately 48 and 40 observations respectively and would therefore not be as greatly influenced by the presence of a few high values.

The sampling procedure (figure 2.2.1, section 2.2.2.2.2.) provided samples from the uncreosoted region of each pole section adjacent to preservative injection sites. The procedure was designed to physically accommodate any differences in injection depth and peripheral separation of injections. Variability existed even between injection sites of the same pole section sample disc, i.e. irrespective of creosote depth, pole diameter and moisture content, and was due to the inability of the injection apparatus, as operated, to provide consistent horizontal injections of identical depth evenly distributed around the pole section. As a consequence, the transverse face of each like numbered wood sample was not identical. Thus, sample position number was an imprecise measure of the distance of a given sample position from the injection site. The interaction of sample position and duration of field exposure (table 2.3.3.8) was therefore a measure of fluoride diffusion over time superimposed on inherent injection differences. However, when these mean fluoride concentrations were combined for the main effects of sample position and field exposure period (table 2.3.3.7), the influence of the injection procedure was lessened. Interaction mean values are therefore good indicators of fluoride concentrations in treated poles up to 20 months after preservative injection, however they do not represent accurate estimates of fluoride diffusion up to 20 months after treatment. For this reason, the main effects of the factors, which in most statistical cases are nullified by a strong interaction between the 2,

cannot be discounted.

2.3.4. The effect of remedial treatment on some wood pole inhabitant micro-organisms.

2.3.4.1. Introduction.

Wood cores removed from the uncreosoted groundline region of Rentex treated 'on-line' distribution poles over a 16 month period (section 2.2.3.2.) were incubated on nutrient agar to allow growth and identification of inhabitant micro-organisms (section 2.2.3.3.1.) for comparison with cores removed from the poles prior to preservative treatment.

2.3.4.2. Results tables and figures.

Tables 2.3.4.1 and 2.3.4.2 indicate initial isolations prior to remedial treatment, and final presence after remedial treatment, of ***Neolentinus lepideus***, bacteria, moulds and zero isolates on wood cores removed from treated and untreated distribution poles respectively. At each indicated core sampling time after treatment (tables 2.3.4.1 and 2.3.4.2), separate representative groups of 20 treated and 20 untreated poles, selected on the basis of percentage moisture content (sections 2.2.3.2.5. and 2.2.3.2.6.), are each separated into 2 groups of 10 'dry' and 10 'wet' poles. Tables 2.3.4.1 and 2.3.4.2 also show mean pole measurements of diameter, height and depth (section 2.2.3.2.1.), moisture content (section 2.2.3.2.3.) and percentage creosote depth (section 2.2.3.3.2.).

Figure 2.3.4.1 shows the original and final percentage of wood cores, from control and remedially treated poles, from which ***N. lepideus***, bacteria and moulds were isolated at each period after remedial treatment. Figure 2.3.4.2 indicates the original and final percentage of cores recovered which were free of microbial growth, and the consequent effect on the

percentage of treated and control poles from which no micro-organisms were isolated at each period after treatment.

Table 2.3.4.3, part A, shows the mean initial and final presence of *N. lepideus*, bacteria and moulds, combined over all sampling periods after treatment, for 'wet' and 'dry' pole groups of remedially treated distribution poles (table 2.3.4.1). Similarly, table 2.3.4.3, part B, indicates these mean values for untreated distribution poles (table 2.3.4.2) and table 2.3.4.3, part C, displays the mean initial and final presence of these organisms on wood cores from treated and control poles irrespective of moisture status.

Oneway analysis of variance was carried out to compare the mean presence of each group of isolated organisms and clear cores (table 2.3.4.3, parts A, B and C) at a number of levels. The P - values of significance for comparisons within treated poles, within control poles and between treated and control poles are presented in tables 2.3.4.4, 2.3.4.5 and 2.3.4.6 respectively.

Table 2.3.4.7, part A, shows the initial and final presence of moulds, identified as strains of *Cladosporium resinae* or *Trichoderma viride*, isolated from treated and control pole cores at 3, 6 and 16 months after remedial treatment. Table 2.3.4.7, part B, indicates the initial and final mean presence of *C. resinae* and *T. viride* in cores from 'wet' and 'dry' treated and control poles. Table 2.3.4.7, part C, displays the mean presence of these moulds for treated and control poles irrespective of moisture status.

2.3.4.3. Distribution pole parameters.

Oneway analysis of variance indicated no significant differences in the mean pole parameter of initial moisture content between 1, 3, 6 and 16 months sampling periods within the groups of treated 'dry' poles, and treated 'wet' poles (table 2.3.4.1). There were also no

Table 2.3.4.1. Initial (1) and final (2) presence of N. lepideus, bacteria and moulds on cores recovered from remedially treated field poles (standard deviations in parenthesis for means of 10).

Pole Moisture Group	Months After Treatment	Mean Pole Moisture (%)		Number of Cores (of 30 recovered) Supporting Growth of:								Mean Pole Diameter (cm)	Mean Pole Creosote (%R)	Mean Pole Depth (m)	Mean Pole Height (m)
				N. lepideus		Bacteria		Moulds		No Isolation					
		1	2	1	2	1	2	1	2	1	2				
'DRY'	1	19.95 (2.22)	21.25 (3.71)	1	4	5	2	23	12	6	15	21.07 (2.95)	52.58 (16.00)	1.55 (0.12)	7.76 (0.62)
	3	20.15 (2.26)	20.80 (2.11)	0	0	8	1	24	8	4	21	21.20 (1.92)	55.78 (17.90)	1.61 (0.14)	8.08 (0.69)
	6	20.25 (2.12)	22.30 (2.71)	0	0	4	2	20	20	9	10	21.70 (3.05)	64.02 (13.14)	1.62 (0.20)	8.12 (0.99)
	16	20.40 (2.21)	21.90 (3.51)	4	1	4	6	19	16	8	13	22.36 (2.92)	56.01 (10.38)	1.64 (0.20)	8.22 (0.99)
'WET'	1	29.00 (6.60)	23.55 (3.95)	2	2	9	5	19	7	6	16	21.95 (2.82)	53.62 (15.47)	1.5 (0.27)	7.51 (1.37)
	3	29.55 (7.24)	29.10 (6.71)	1	0	14	4	17	7	5	19	21.92 (3.16)	57.56 (14.09)	1.55 (0.14)	7.76 (0.74)
	6	30.20 (7.36)	36.10 (12.82)	2	1	15	3	20	18	3	11	19.58 (1.28)	54.93 (18.43)	1.51 (0.12)	7.56 (0.61)
	16	30.85 (8.48)	33.95 (17.18)	2	0	10	10	22	15	2	14	21.10 (2.57)	59.37 (14.99)	1.52 (0.13)	7.61 (0.64)

Table 2.3.4.2. Initial (1) and final (2) presence of N. lepideus, bacteria and moulds on cores recovered from untreated field poles (standard deviations in parenthesis for means of 10).

Pole Moisture Group	Months After Treatment	Mean Pole Moisture (%)		Number of Cores (of 30 recovered) Supporting Growth of:								Mean Pole Diameter (cm)	Mean Pole Creosote (%R)	Mean Pole Depth (m)	Mean Pole Height (m)
				N.lepideus		Bacteria		Mould		No Isolation					
		1	2	1	2	1	2	1	2	1	2				
'DRY'	1	20.25 (3.69)	20.55 (2.34)	1	0	7	5	23	24	4	5	22.32 (2.58)	48.75 (15.58)	1.62 (0.16)	8.12 (0.80)
	3	20.65 (2.89)	21.05 (1.95)	2	3	4	6	24	18	6	11	21.01 (3.42)	61.02 (16.89)	1.53 (0.11)	7.66 (0.54)
	6	20.80 (2.85)	21.10 (2.38)	2	2	7	0	24	26	5	4	22.14 (2.43)	51.84 (13.08)	1.58 (0.20)	7.87 (0.97)
	16	21.15 (2.65)	20.00 (2.36)	2	2	6	12	20	15	9	12	23.00 (1.50)	58.44 (16.74)	1.56 (0.18)	7.81 (0.91)
'WET'	1	31.75 (7.93)	26.50 (4.02)	0	0	13	15	19	16	6	7	21.48 (1.95)	53.81 (13.96)	1.54 (0.13)	7.72 (0.67)
	3	32.25 (8.64)	26.80 (3.61)	0	0	12	14	15	8	6	11	20.63 (1.74)	60.32 (17.96)	1.5 (0.14)	7.51 (0.71)
	6	32.80 (8.60)	27.65 (6.84)	0	2	12	3	18	27	4	3	20.29 (1.40)	56.57 (11.16)	1.51 (0.12)	7.56 (0.61)
	16	33.30 (9.77)	33.60 (17.86)	5	3	14	15	16	13	4	8	20.17 (1.95)	57.08 (8.71)	1.46 (0.09)	7.32 (0.45)

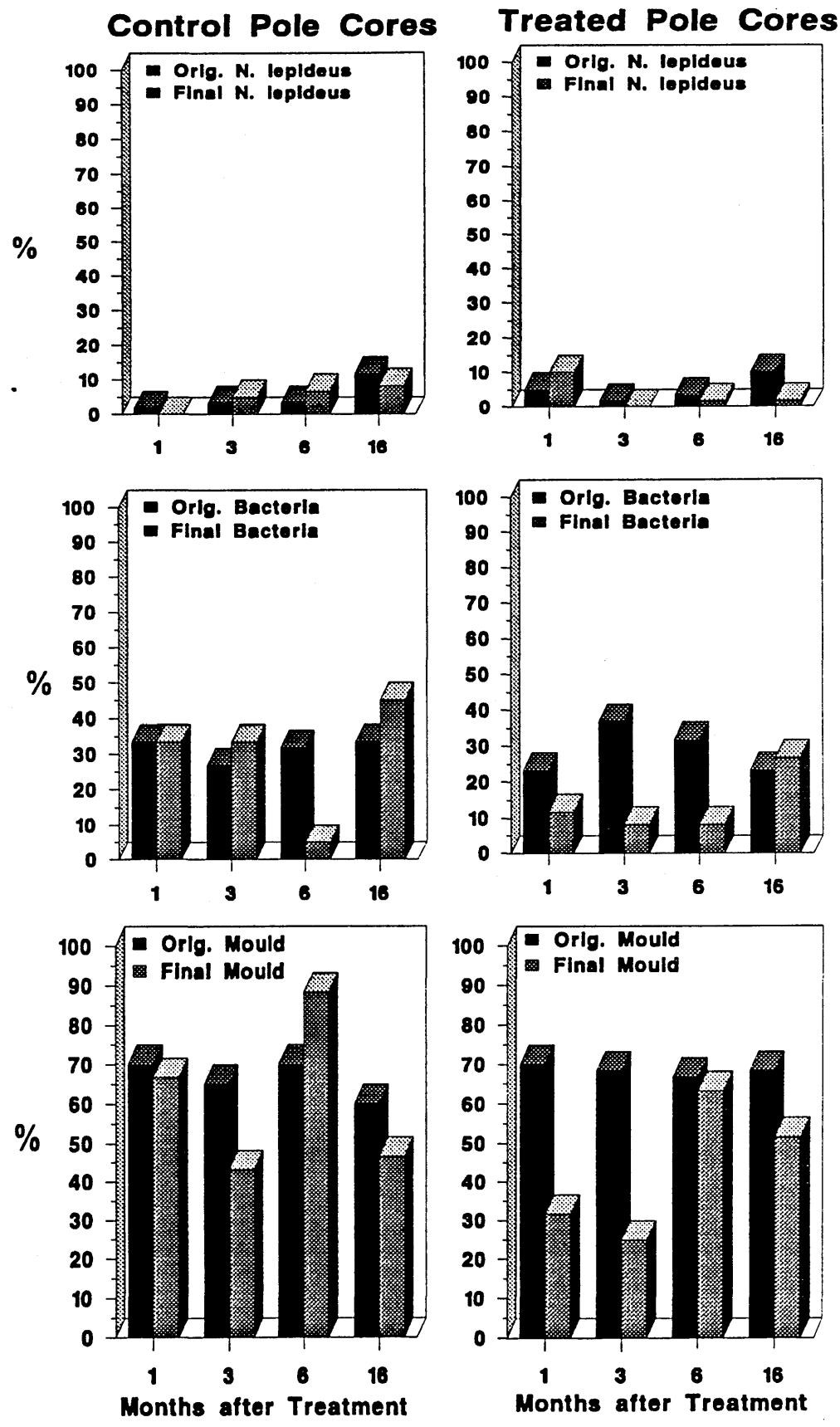


Figure 2.3.4.1. The original and final percentages of wood cores, from control and remedially treated creosoted distribution poles, from which *N. lepideus*, bacteria and moulds were isolated at 1, 3, 6 and 16 months after remedial treatment. All original cores were recovered from poles prior to remedial treatment.

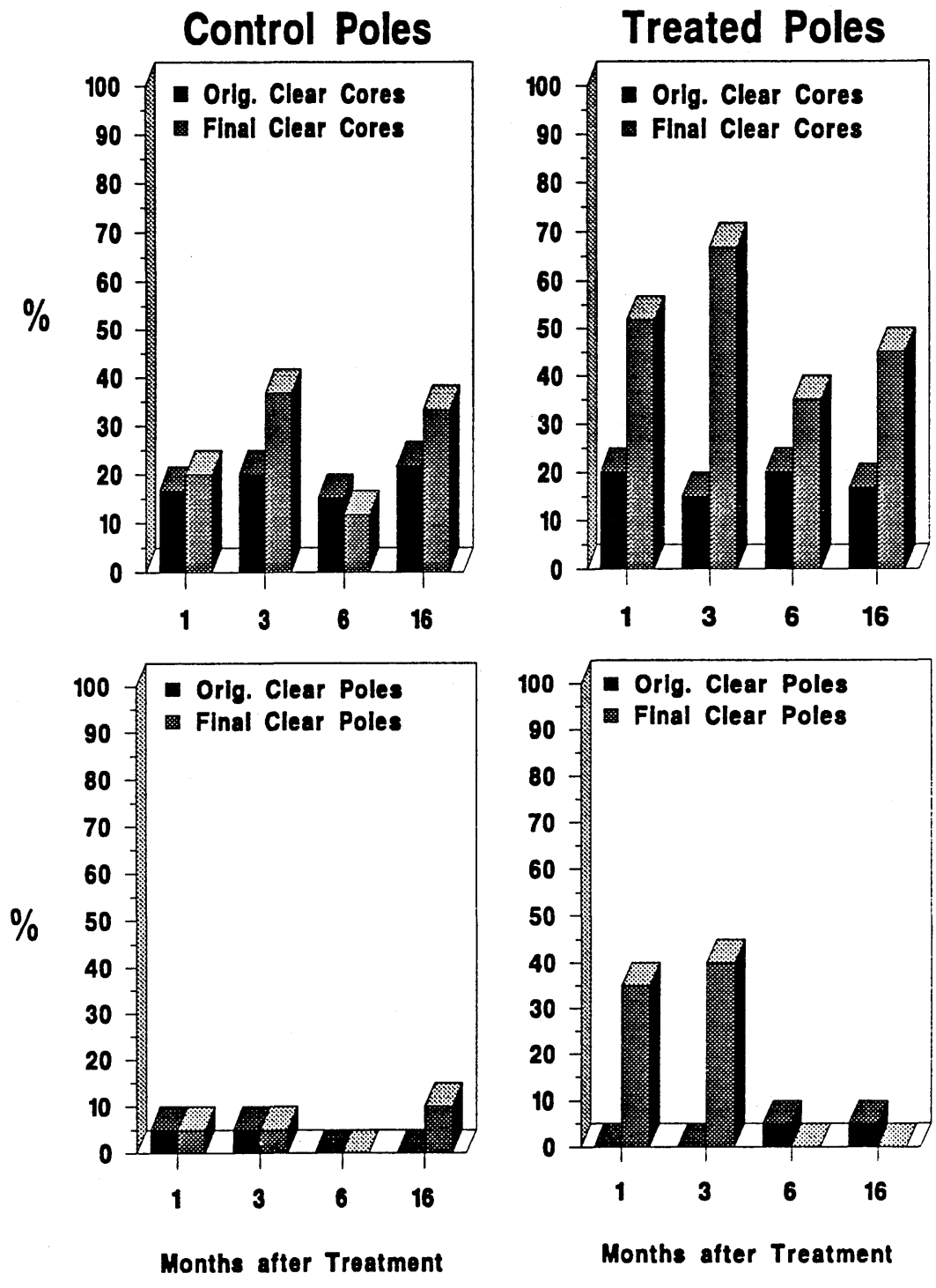


Figure 2.3.4.2. The original and final percentages of wood cores from control and remedially treated creosoted distribution poles which were free of microbial growth at 1, 3, 6 and 16 months after remedial treatment, and the consequent percentage of these poles from which no micro-organisms were isolated. All original cores were recovered from poles prior to remedial treatment.

Table 2.3.4.3. Mean initial (1) and final (2) presence of N. lepideus, bacteria, moulds and zero isolates, on wood cores, combined over all sampling periods from:

A - Remedially treated poles of high and low moisture content

B - Control poles of high and low moisture content

C - Remedially treated and control poles

(Standard deviations in parenthesis are for means of 4 (A, B) and 8 (C)].

	Pole Group	Moisture Status	Mean Number of Cores Supporting Growth of:							
			N. lepideus		Bacteria		Mould		No Isolation	
			1	2	1	2	1	2	1	2
A	Treated	'Dry'	1.25 (1.89)	1.25 (1.89)	5.25 (1.89)	2.75 (2.22)	21.50 (2.37)	14.00 (5.16)	6.75 (2.22)	14.75 (4.65)
	Treated	'Wet'	1.75 (0.51)	0.75 (0.96)	12.00 (2.94)	5.50 (3.12)	19.50 (2.07)	11.75 (5.61)	4.00 (0.99)	15.00 (3.36)
B	Control	'Dry'	1.75 (0.51)	1.75 (1.26)	6.00 (1.41)	5.75 (4.92)	22.75 (1.89)	20.75 (5.13)	6.00 (2.16)	8.00 (4.08)
	Control	'Wet'	1.25 (2.49)	1.25 (1.50)	12.75 (0.96)	11.75 (5.85)	17.00 (1.83)	16.00 (8.04)	5.00 (1.14)	7.25 (3.30)
C	Treated	D/W	1.50 (1.31)	1.00 (1.41)	8.62 (4.27)	4.12 (2.90)	20.50 (2.33)	12.88 (5.14)	5.38 (2.39)	14.88 (3.76)
	Control	D/W	1.50 (1.69)	1.50 (1.31)	9.38 (3.78)	8.75 (5.95)	19.88 (3.52)	18.38 (6.74)	5.50 (1.69)	7.62 (3.46)

Table 2.3.4.4. Significance table for statistical comparisons within treated poles.

Comparison (oneway analysis of variance)	*N. lepideus*	Bacteria	Moulds	Clear Cores
'Wet' x 'dry' initial presence	x	0.008	x	x
'Wet' x 'dry' final presence	x	x	x	x
'Wet' initial x final presence	x	0.023	0.041	0.001
'Dry' initial x final presence	x	x	0.039	0.021
Initial x final presence	x	0.027	0.002	0.0005

Table 2.3.4.5. Significance table for statistical comparisons within control poles.

Comparison (oneway analysis of variance)	*N. lepideus*	Bacteria	Moulds	Clear Cores
'Wet' x 'dry' initial presence	x	0.0005	0.005	x
'Wet' x 'dry' final presence	x	0.015	x	x
'Wet' initial x final presence	x	x	x	x
'Dry' initial x final presence	x	x	x	x
Initial x final presence	x	x	x	x

Table 2.3.4.6. Significance table for statistical comparisons between treated and control poles.

Comparison (oneway analysis of variance)	*N. lepideus*	Bacteria	Moulds	Clear Cores
'Wet' initial presence, T x C	x	x	x	x
'Dry' initial presence, T x C	x	x	x	x
'Dry' final presence, T x C	x	x	x	x
'Wet' final presence, T x C	x	0.002	x	0.017
Initial presence, T x C	x	x	x	x
Final presence, T x C	x	0.020	x	0.001

X = No significant difference.

Table 2.3.4.7. A - Initial (1) and final (2) presence of C. resinae and T. viride strains in treated and control pole cores of high and low moisture content at 3, 6 and 16 months after remedial treatment.
B - Mean initial and final presence of C. resinae and T. viride strains in treated and control pole cores of high and low moisture content combined over all sampling periods.
C - Mean initial and final presence of C. resinae and T. viride strains in treated and control pole cores.
[Standard deviations in parenthesis for means of 3 (B) and 6 (C)]

	Months After Treatment 'Wet'/'Dry'	Treated poles				Control poles			
		Number of Cores with Mould Growth displaying Growth of:							
		C. resinae		*T. viride*		*C. resinae*		*T. viride*	
		1	2	1	2	1	2	1	2
A	3 D	17	5	2	0	19	10	1	1
	6 D	14	10	2	9	21	19	1	11
	16 D	13	6	1	1	16	10	2	0
	3 W	15	2	2	0	11	5	1	1
	6 W	13	7	4	7	10	9	2	9
	16 W	19	5	3	0	11	5	1	0
B	D	14.67	7.00	1.67	3.33	18.67	13.00	1.33	4.00
		(2.08)	(2.64)	(0.58)	(4.93)	(2.52)	(5.20)	(0.58)	(6.08)
	W	15.67	4.67	3.00	2.33	10.67	6.33	1.33	3.33
		(3.06)	(2.52)	(1.00)	(4.04)	(0.58)	(2.30)	(0.58)	(4.93)
C	D/W	15.17	5.83	2.33	2.83	14.67	9.67	1.33	3.67
		(2.40)	(2.65)	(1.03)	(4.07)	(4.68)	(5.10)	(0.52)	(4.97)

significant differences for identical separate comparisons of final moisture content, pole diameter, percentage creosote depth, pole depth and pole height within these treated pole groups (tables 2.3.4.1). Similarly, there were no significant differences for identical separate comparisons of all these pole parameters within the groups of control 'dry' poles and control 'wet' poles (table 2.3.4.2). There were also no significant differences when comparisons of these pole parameters were made between control and treated 'dry' poles or control and treated 'wet' poles at 1, 3, 6 and 16 months (tables 2.3.4.1. and 2.3.4.2.).

However, the initial mean moisture content of 'dry' control and treated poles was very significantly lower, at each sampling time, than the initial mean moisture content of 'wet' control and treated poles respectively, $P< 0.0005$ and 0.0005. Identical significant differences were found between final mean moisture contents for both control and treated poles (tables 2.3.4.1. and 2.3.4.2.).

The procedure for selection of poles (section 2.2.3.2.5.), for removal of a 2nd set of core samples (section 2.2.3.2.6.), was therefore successful in providing comparative groups of 'wet' and 'dry', control and treated poles.

2.3.4.4. Isolation of micro-organisms.

2.3.4.4.1. General trends indicated by tables and figures.

Tables 2.3.4.1. and 2.3.4.2. clearly indicate that moulds made up the larger portion of isolated organisms irrespective of preservative application or sample time. The greater isolation of bacteria in poles of high moisture content from either group of control or treated poles is also shown, as is the infrequent isolation of the basidiomycete *N. lepideus*. Remedial treatment appeared to cause a decline in the presence of bacteria and moulds especially at 1 and 3 months after treatment irrespective of the moisture status of poles

(table 2.3.4.1). Treatment also appeared to result in more wood cores from which no micro-organisms were isolated.

Figure 2.3.4.1, like tables 2.3.4.1 and 2.3.4.2, displays the preponderance of moulds in original and final cores from treated and control poles and again the infrequent presence of ***N. lepideus***. Examination of original isolation percentages for both treated and control pole cores (figure 2.3.4.1) indicates the generally stable proportions of moulds, in particular, and bacteria, when cores were removed at the same date, December 1989. Comparison of the original and final isolation percentages of bacteria and moulds for control poles (figure 2.3.4.1) indicates a seasonal variation in pole populations of organisms. For instance, 6 months after treatment, in July 1990, the percentage of isolated organisms which were moulds rose for control poles. Conversely, isolated bacteria percentages fell at this time, which supports the indications of a preference for 'wetter' conditions given in tables 2.3.4.1 and 2.3.4.2. The effects of remedial treatment appeared to be superimposed on this seasonal population variation, with apparent reductions in percentage re-isolations of moulds and bacteria, in treated poles, up to 6 months after treatment (figure 2.3.4.1). The low isolation of ***N. lepideus*** within the poles studied prevents meaningful comment on treatment effects on this organism.

Figure 2.3.4.2 indicates the distinct percentage increase in cores displaying no growth for treated poles particularly at 1 and 3 months after treatment. Again, comparison between treated and control clear core percentages indicates that the treatment effect is superimposed on a seasonal population effect. The effects of remedial treatment on the percentage of poles sampled which were apparently free of microbial growth are clearly shown at 1 and 3 months after treatment (figure 2.3.4.2). Thereafter the treatment appears to have no beneficial effect.

2.3.4.4.2. Isolation of *N. lepideus.*

Tables 2.3.4.4, 2.3.4.5 and 2.3.4.6 show the lack of significant differences for the mean presence of ***N. lepideus*** (table 2.3.4.3, parts A, B and C) for any comparison. However, to base a conclusion, as to the effect of Rentex treatment on ***N. lepideus***, would be unwise given the infrequent and erratic occurrence of this organism (tables 2.3.4.1, 2.3.4.2, 2.3.4.3 and figure 2.3.4.1).

2.3.4.4.3. Isolation of bacteria.

The initial presence of bacteria was significantly higher in 'wet' poles than in 'dry' for both treated and control pole groups (tables 2.3.4.4, 2.3.4.5 and 2.3.4.3, parts A and B) confirming the preference of bacteria for an environment of higher moisture content (section 2.3.4.4.1.). There was no significant difference in the initial presence of these organisms, between treated and control poles (tables 2.3.4.6 and 2.3.4.3, part C) indicating the stability of populations prior to remedial treatment. The lack of significant differences between initial and final presence of bacteria in control poles (tables 2.3.4.5 and 2.3.4.3, part C) ensured that no significant consistent seasonal effect occurred to mask the effects of the remedial treatment. The final presence of these organisms in 'wet' and 'dry' poles was not significantly different within treated poles (tables 2.3.4.4 and 2.3.4.3, part A), therefore the treatment had nullified the normal population imbalance between 'wet' and 'dry', which still existed in control poles (tables 2.3.4.5 and 2.3.4.3, part B). This effect of the treatment was confirmed by the significantly lower final presence of these organisms in 'wet' treated poles compared with 'wet' control poles (tables 2.3.4.6 and 2.3.4.3, parts A and B), which gave an overall significant reduction between initial and final presence of bacteria in treated poles (tables 2.3.4.4 and 2.3.4.3, part A). Due to the treatment effect in 'wet' treated poles (tables 2.3.4.4 and 2.3.4.3, part A) and the lack of a significant seasonal effect in control poles (tables 2.3.4.5 and 2.3.4.3, part C), the final mean presence of bacteria was significantly reduced in treated poles compared with control poles (tables 2.3.4.6 and 2.3.4.3, part C).

2.3.4.4.4. Isolation of moulds.

The mean initial 'wet' and 'dry' presence respectively of moulds in treated poles was significantly higher than their mean final presence (tables 2.3.4.4 and 2.3.4.3, part A) giving a significantly lower final mean presence overall within treated poles (tables 2.3.4.4 and 2.3.4.3, part C). The lack of a significant difference in the final mean presence of moulds between control and treated poles (tables 2.3.4.6 and 2.3.4.3, part C) indicated that a seasonal effect was operating, which, though it did not obscure differences between the combined final mean values for moulds (table 2.3.4.3, part C) it did serve to increase variation particularly for the final mean value of moulds in 'wet' control poles (table 2.3.4.3, part B) in the absence of remedial treatment. This, combined with the significantly lower mean initial presence of moulds in 'wet' control poles (tables 2.3.4.3, part B and 2.3.4.5.) which was reflected in the final mean presence of moulds in such poles (table 2.3.4.3, part B) resulted in no significant differences between the final mean presence of moulds in control and treated poles (tables 2.3.4.6 and 2.3.4.3, part C).

2.3.4.4.5. Wood cores displaying no growth of micro-organisms.

Tables 2.3.4.4 and 2.3.4.3, part A, indicate that the final mean occurrence of clear cores was significantly greater than the initial mean occurrence at each level of comparison within treated poles. The levels of significance (table 2.3.4.4) indicate that the treatment was more effective at higher moisture contents and this is confirmed by the significantly greater occurrence of final clear cores between 'wet' treated and control poles (tables 2.3.4.6 and 2.3.4.3 parts A and B). The mean final occurrence of clear cores in treated poles over all moisture contents was significantly greater than control pole values (tables 2.3.4.6 and 2.3.4.3, part C). As expected, no significant differences were found within control poles (table 2.3.4.5). Therefore, remedial treatment did produce a consistent significant reduction in numbers of isolated micro-organisms over the 16 months of the study.

2.3.4.4.6. Demographic analysis of common mould isolates.

To examine the effect of remedial treatment on moulds, the most common organisms isolated (tables 2.3.4.1. and 2.3.4.2.), the presence of strains of the moulds ***Cladosporium resinae*** and ***Trichoderma viride*** at 3, 6 and 16 months after remedial treatment was separately noted (table 2.3.4.7, part A). ***C. resinae*** was the most commonly isolated mould as is indicated by comparison of the initial presence of this mould in treated and control poles (table 2.3.4.7, part A) respectively with the presence of moulds generally in tables 2.3.4.1 and 2.3.4.2.

The relative final presence of both ***C. resinae*** and ***T. viride*** rose at the 6 month sampling period irrespective of remedial treatment (table 2.3.4.7, part A). This confirmed the indications given of an increase in isolation of moulds generally in the summer (section 2.3.4.4.1.). Whereas final ***T. viride*** re-isolations were greater than initial levels (cores removed in winter), this was not the case for ***C. resinae***. This suggests that the extent of ***T. viride*** colonisation in distribution poles was largely dictated by temperature alone even after remedial treatment. However, the presence of ***C. resinae*** may be affected by competition with other organisms as well as remedial treatment.

Oneway analysis of variance indicated that the mean initial or final presence of *C. **resinae*** was not significantly different between 'wet' and 'dry' treated pole groups (table 2.3.4.7, part B). However, the mean final presence of ***C. resinae*** was significantly lower than its initial presence in both 'wet' and 'dry' treated pole groups, $P = 0.009$ and 0.017 respectively (table 2.3.4.7, part B). In consequence, a very highly significant fall in the mean final presence of this organism, over all moisture contents, was found for treated poles, $P < 0.0005$ (table 2.3.4.7, part C).

Though there was no significant difference between the mean initial and final presence of *C. **resinae*** in 'dry' control poles, the final mean presence of ***C. resinae*** in 'wet' control

poles was significantly lower, P = 0.034, than its initial presence (table 2.3.4.7, part B). The significantly lower initial mean presence of *C. resinae* in 'wet' control poles, compared with 'dry' control poles, P = 0.006, and the low mean final presence of this organism in 'wet' control poles (table 2.3.4.7, part B) would tend to indicate a preference for dry conditions. This follows a significant trend shown for the initial mean presence of moulds generally, in control poles, P = 0.005 (section 2.3.4.3.4., tables 2.3.4.3 part B and 2.3.4.5). There was no significant difference between initial and final mean presence of *C. resinae*, in control poles combined for all moisture contents (table 2.3.4.7, part C).

Statistical comparisons between treated and control groups indicated that the initial mean presence of *C. resinae* in 'wet' control poles was just significantly lower than that for 'wet' treated poles, P = 0.050 (table 2.3.4.7, part B). This lower mean presence, was reflected in the final mean presence of *C. resinae* in 'wet' control poles (table 2.3.4.7, part B) and effectively served to remove any differences in final mean presence of *C. resinae* between treated and control pole groups over all moisture contents (table 2.3.4.7, part C).

Given that there was no significant difference between the initial and final mean presence of *C. resinae* in control poles, the very significantly lower final mean presence of this mould compared to its initial mean presence, in treated poles, does appear to be due to remedial treatment. The lack of significant differences between control and treated pole final mean presence of *C. resinae* demonstrates how strong statistical indications of a preservative effect can be obscured by natural variations of micro-organism populations in distribution poles.

Statistical comparisons of means for the presence of *T. viride* produced no meaningful results (table 2.3.4.7, parts B and C). This was due to the limited initial presence of this organism in both pole groups and the variability in final mean presence (table 2.3.4.7, parts B and C) as a result of the dramatic increase in occurrence of this mould at the 6 month sampling period (table 2.3.4.7, part A). It would appear that a seasonal temperature effect

was the most powerful influencing factor for ***T. viride*** presence in both pole groups, though remedial treatment may have 'damped' this effect slightly (table 2.3.4.7, part A).

2.3.5. Fluoride and chromium concentrations in cores from remedially treated 'in service' distribution poles.

2.3.5.1. Results tables.

The mean fluoride and chromium concentrations in wood cores recovered from each of 11 'on-line' distribution poles 18 months after remedial treatment (section 2.2.4.) is shown in table 2.3.5.1. Pole parameters are shown in table 2.3.5.2.

2.3.5.2. Statistical determinations of the influence of pole parameters on fluoride and chromium levels in treated poles.

2.3.5.2.1. Fluoride.

A stepwise regression of each preservative element concentration in distribution poles (table 2.3.5.1) was carried out on 6 pole parameters (table 2.3.5.2) in order to determine the influence of these parameters on concentrations of fluoride and chromium (section 2.3.5.2.2.) remaining in poles 18 months after remedial treatment. By this statistical analysis all factors which were found to be unrelated to mean preservative element levels were automatically excluded from the analysis. A predictive equation for mean percentage fluoride content of cores was as follows:

$$\mathbf{F\ \% = 0.0586 + (0.00547 \times Moisture\ 2) - (0.0153 \times Pole\ height)}$$

The factors of pole moisture content 2 and pole height were significant at $P < 0.0005$ and $P = 0.001$ respectively. The coefficient of determination indicated that the equation explained 72.2 % of the variance. Analysis of variance showed the model as very highly

Table 2.3.5.1. Mean fluoride and chromium concentration (% w/w) in wood cores recovered from distribution poles 18 months after remedial treatment (standard deviations* and standard errors** in parenthesis for means of 2 and 22 respectively).

Pole Number	Mean Fluoride Concentration in Core Group	Mean Chromium Concentration in Core Group	Mean Fluoride Conc.	Mean Chromium Conc.
	*	*	**	**
1	0.0273 (0.0016)	0.0312 (0.0006)		
2	0.0476 (0.0007)	0.0323 (0.0008)		
3	0.0578 (0.0024)	0.0308 (0.0008)		
4	0.0574 (0.0016)	0.0312 (0.0004)		
5	0.0571 (0.0008)	0.0412 (0.0007)	0.0595	0.0285
6	0.1152 (0.0119)	0.0582 (0.0010)	(0.0063)	(0.0028)
7	0.1150 (0.0034)	0.0267 (0.0010)		
8	0.0333 (0.0005)	0.0101 (0.0005)		
9	0.0638 (0.0046)	0.0134 (0.0009)		
10	0.0301 (0.0013)	0.0216 (0.0005)		
11	0.0496 (0.0019)	0.0165 (0.0007)		

Table 2.3.5.2. Initial (1) and final (2) parameters of distribution poles from which wood cores were recovered (standard deviations in parenthesis for means of 11).

Pole Number	Moisture Content (%)		Diameter (cm)	Pole Depth (m)	Pole Height (m)	Creosote Depth (%)
	1	2				
1	17.0	21.0	20.7	1.73	8.67	62.8
2	19.0	18.0	21.0	1.63	8.17	52.4
3	20.0	23.0	22.9	1.63	8.17	52.4
4	20.0	19.0	21.0	1.22	6.08	28.6
5	21.0	20.0	21.0	1.42	7.08	47.6
6	21.0	30.0	19.1	1.32	6.58	57.6
7	24.0	24.0	20.4	1.52	7.58	53.9
8	25.0	20.0	25.8	1.73	8.67	69.8
9	26.5	25.0	24.8	1.63	8.17	24.6
10	27.5	20.0	25.5	1.73	8.67	66.7
11	28.5	24.0	20.1	1.73	8.67	74.6
Means	22.7	22.2	22.0	1.57	7.86	53.7
	(3.80)	(3.46)	(2.34)	(0.18)	(0.92)	(15.7)

significant, $P < 0.0005$ and only the findings for cores from pole 7 (table 2.3.5.1.) deviated from the model prediction, i.e. 9.1 % of the total observations were greater than predicted values.

Therefore, a higher final pole moisture content was associated with an increase in the concentration of fluoride in these distribution poles, whereas a reduction in fluoride concentration was associated with increasing pole height.

2.3.5.2.2. Chromium.

A predictive equation for mean percentage chromium content of cores was as follows:

$$\text{Cr \%} = 0.0981 - (0.0458 \times \text{Pole depth}) - (0.00195 \times \text{Moisture 1}) + \\ + (0.00133 \times \text{Moisture 2}) + (0.000319 \times \text{\% Creosote depth})$$

The factors of pole depth, initial moisture content, final moisture content and percentage creosote depth were significant at $P < 0.0005$, $P < 0.0005$, $P = 0.006$ and $P = 0.009$ respectively. The equation explained 82.8 % of the variance. The model was very highly significant at $P < 0.0005$ and none of the observations differed substantially from the predicted values.

Therefore, higher concentrations of chromium in these distribution poles were associated with decreasing pole depth, or lower initial pole moisture content, or higher final pole moisture content, or a deeper penetration of creosote, when all other pole parameters remained stable. Lower concentrations of this preservative element were associated with contra-indications for each of these pole parameters when all other pole parameters remained stable.

2.3.6. Fluoride and chromium concentrations in soils adjacent to Rentex treated 'on-line' distribution poles

2.3.6.1. Introduction.

Soil samples recovered from positions in close proximity to 14 Rentex treated field poles at 1 week, 1 month, 6 and 12 months after remedial treatment (sections 2.2.5.2. and 2.2.5.3.), were analysed for fluoride and chromium content (section 2.2.5.4.) for comparison with background values of these elements.

2.3.6.2. Soil analysis data.

Tables 2.3.6.1, 2.3.6.2 and 2.3.6.3 show fluoride and chromium concentrations in soil samples recovered from downslope of remedially treated distribution poles (table 2.3.6.4) at 6 cm and 25 cm and for a background soil sample > 50 m away respectively, at 1 week, 1 month, 6 and 12 months after preservative application.

Table 2.3.6.5 indicates fitted mean soil concentrations of fluoride and chromium for the statistically significant main effects of time after pole treatment and sample distance from the pole and their interaction.

Table 2.3.6.6 indicates fitted mean soil concentrations of fluoride and chromium for the statistically significant main effects of time after pole treatment and sample distance from the pole and their interaction, when analysis of variance excluded concentrations of both preservative elements in background soil samples > 50 m distant from poles.

Table 2.3.6.1. Mean soil concentrations of A, fluoride and B, chromium, 6 cm downslope of remedially treated distribution poles at 1 week, 1, 6 and 12 months after treatment (standard deviations in parenthesis for means of 2).

A				
Soil Sample	Mean Soil Concentration of Fluoride (ug/g) after:			
Pole No.	1 Week (1)	1 Month (2)	6 Months (3)	12 Months (4)
1	3422.50 (048.00)	3481.00 (630.00)	1649.40 (130.20)	1280.60 (052.50)
2	0722.96 (004.02)	1720.60 (000.50)	1417.00 (013.10)	2124.10 (118.60)
3	0498.00 (032.20)	2661.20 (071.30)	1163.40 (054.50)	1857.10 (073.30)
4	0933.40 (034.90)	1072.40 (054.80)	1326.40 (039.30)	1218.30 (037.40)
5	0431.70 (060.90)	0372.97 (000.53)	1593.60 (112.90)	1261.90 (051.90)
6	0461.13 (002.19)	0548.05 (003.28)	0972.50 (008.89)	1192.30 (052.90)
7	0731.07 (001.59)	2095.40 (096.30)	2195.70 (064.30)	4829.60 (138.50)
8	0748.80 (033.80)	0562.87 (002.97)	0725.00 (016.70)	0745.50 (100.80)
9	0709.20 (041.00)	0795.13 (000.80)	1327.60 (049.10)	0409.07 (012.73)
10	2533.00 (300.00)	0618.10 (014.60)	2833.10 (102.10)	0717.40 (064.30)
11	3601.70 (020.80)	0360.71 (002.49)	0522.00 (019.40)	0753.30 (034.30)
12	0599.87 (004.76)	1270.50 (085.80)	2168.30 (086.30)	1919.20 (118.60)
13	4233.50 (116.40)	1518.30 (005.80)	0702.60 (046.20)	1199.00 (072.20)
14	0680.27 (000.90)	0673.41 (004.28)	0737.78 (008.60)	0785.80 (034.40)

B				
Soil Sample	Mean Soil Concentration of Chromium (ug/g) after:			
Pole No.	1 Week (1)	1 Month (2)	6 Months (3)	12 Months (4)
1	0280.50 (016.00)	0244.25 (008.74)	0110.25 (000.67)	0171.73 (007.29)
2	0677.20 (015.60)	0369.59 (009.28)	0236.13 (008.82)	0275.50 (004.44)
3	0144.20 (001.62)	0434.50 (007.35)	0298.98 (002.07)	0343.48 (005.64)
4	0346.75 (001.12)	0195.45 (002.25)	0268.42 (004.41)	0218.96 (004.80)
5	0094.38 (009.07)	0089.04 (002.74)	0182.88 (004.23)	0086.19 (004.32)
6	0232.86 (001.44)	0143.03 (002.39)	0137.76 (004.72)	0111.62 (000.35)
7	0233.77 (003.66)	0587.60 (021.40)	0317.17 (003.37)	0451.29 (003.72)
8	0685.87 (005.52)	0149.04 (013.45)	0229.70 (035.70)	0118.33 (010.73)
9	LOST	0107.18 (010.91)	0200.15 (005.73)	0099.85 (003.59)
10	0761.70 (007.65)	0226.46 (005.18)	0244.17 (006.96)	0117.13 (006.68)
11	1707.00 (027.80)	0152.10 (004.26)	0174.25 (006.21)	0051.25 (005.07)
12	0261.99 (005.04)	0323.59 (004.10)	0252.99 (001.74)	0338.68 (007.70)
13	0986.17 (004.41)	0345.99 (004.94)	0129.19 (004.81)	0359.49 (004.67)
14	0218.46 (008.83)	0189.35 (009.57)	0170.85 (001.70)	0075.99 (003.90)

Table 2.3.6.2. Mean soil concentrations of A, fluoride and B, chromium, 25 cm downslope of remedially treated distribution poles at 1 week, 1, 6 and 12 months after treatment (standard deviations in parenthesis for means of 2).

A				
Soil Sample	Mean Soil Concentration of Fluoride (ug/g) after:			
Pole No.	1 Week (1)	1 Month (2)	6 Months (3)	12 Months (4)
1	0274.91 (001.47)	0456.20 (029.80)	0690.80 (017.90)	0344.90 (015.50)
2	0624.90 (036.30)	0488.00 (035.30)	0753.37 (004.67)	0276.12 (000.94)
3	0344.45 (000.00)	0614.00 (146.00)	0273.57 (008.46)	0314.85 (005.59)
4	0597.90 (107.80)	0470.31 (001.00)	0750.30 (030.90)	0315.80 (019.70)
5	0263.40 (024.20)	0466.49 (002.51)	0499.50 (023.40)	0543.00 (017.90)
6	0415.50 (030.40)	0385.07 (004.02)	0638.90 (015.60)	0331.80 (015.00)
7	0395.59 (000.18)	0509.20 (033.10)	0809.50 (014.30)	0768.97 (005.73)
8	0270.77 (001.34)	0376.80 (002.10)	1052.60 (121.50)	0358.33 (011.29)
9	0259.80 (022.90)	0393.77 (003.22)	0571.43 (010.72)	0456.60 (030.80)
10	0209.40 (031.90)	0341.51 (001.40)	0714.30 (004.53)	0196.77 (006.99)
11	0345.64 (002.86)	0394.66 (001.50)	0267.33 (005.07)	0134.46 (011.21)
12	0336.86 (002.02)	0383.27 (002.56)	0653.50 (026.00)	0297.30 (017.6)
13	0382.88 (001.04)	0573.50 (139.70)	0717.10 (025.90)	0256.27 (010.77)
14	0272.19 (001.59)	0352.86 (000.41)	0806.60 (015.30)	0116.33 (003.84)

B				
Soil Sample	Mean Soil Concentration of Chromium (ug/g) after:			
Pole No.	1 Week (1)	1 Month (2)	6 Months (3)	12 Months (4)
1	0115.82 (013.42)	0087.15 (005.32)	0127.33 (004.60)	0056.66 (004.41)
2	0176.50 (000.67)	0218.59 (005.81)	0201.90 (001.05)	0140.71 (003.88)
3	LOST	0113.58 (003.17)	0095.03 (002.86)	0035.48 (006.66)
4	0233.75 (005.02)	0158.73 (001.05)	0183.82 (002.04)	0067.63 (002.31)
5	0117.18 (006.17)	0883.83 (007.39)	0115.14 (004.57)	0088.62 (000.33)
6	0114.75 (004.79)	0099.69 (002.45)	0123.33 (001.44)	0100.50 (003.29)
7	0185.46 (005.29)	0220.71 (007.16)	0201.72 (001.81)	0147.32 (004.69)
8	0058.72 (001.05)	0100.00 (000.33)	0168.34 (002.13)	0109.69 (004.86)
9	0087.73 (010.65)	0144.07 (006.40)	0153.10 (001.44)	0066.16 (003.90)
10	0093.22 (010.47)	0101.99 (003.78)	0134.11 (012.29)	0106.66 (007.28)
11	0142.21 (011.96)	0081.19 (006.04)	0137.68 (006.79)	0032.22 (003.17)
12	0144.21 (007.96)	0130.79 (009.04)	0148.39 (003.63)	0095.58 (003.64)
13	0111.33 (001.03)	0143.73 (012.77)	0109.83 (006.30)	0064.12 (006.42)
14	0125.72 (006.80)	0076.28 (005.61)	0167.75 (005.93)	0046.56 (005.86)

Table 2.3.6.3. Mean soil concentrations of A, fluoride and B, chromium, > 50 m upslope of remedially treated distribution poles at 1 week, 1, 6 and 12 months after treatment (standard deviations in parenthesis for means of 2).

A				
Soil Sample	Mean Soil Concentration of Fluoride (ug/g) after:			
Pole No.	1 Week (1)	1 Month (2)	6 Months (3)	12 Months (4)
1	0075.21 (001.92)	0296.63 (006.55)	0264.90 (025.50)	LOST
2	0252.80 (034.10)	0436.70 (030.00)	0322.52 (009.51)	0352.00 (015.80)
3	0351.20 (028.70)	0449.20 (015.00)	0397.60 (026.40)	0335.62 (008.24)
4	0269.20 (026.90)	0331.40 (000.99)	0244.80 (015.50)	0170.10 (014.60)
5	0338.50 (062.00)	0373.90 (018.90)	0228.50 (018.40)	0219.10 (016.20)
6	0311.70 (070.80)	0404.40 (046.50)	0648.67 (000.38)	0346.64 (008.87)
7	0373.90 (027.90)	0488.02 (000.98)	0247.90 (019.80)	0231.60 (001.11)
8	0231.83 (002.73)	0190.05 (000.47)	0232.10 (026.20)	0189.05 (002.29)
9	0165.00 (016.20)	0313.87 (000.52)	0323.80 (016.60)	0233.60 (028.20)
10	0259.00 (053.40)	0167.62 (000.00)	0140.12 (012.53)	0182.52 (008.30)
11	0130.00 (045.40)	0301.82 (002.02)	0302.20 (026.00)	0171.22 (007.17)
12	0247.36 (000.32)	0328.16 (004.82)	0199.95 (006.01)	0235.50 (018.00)
13	0311.10 (051.10)	0368.17 (001.91)	0471.40 (031.90)	0491.00 (018.20)
14	0230.70 (018.80)	0314.60 (018.10)	0302.94 (008.40)	0247.92 (010.51)

B				
Soil Sample	Mean Soil Concentration of Chromium (ug/g) after:			
Pole No.	1 Week (1)	1 Month (2)	6 Months (3)	12 Months (4)
1	0075.66 (003.49)	0054.46 (002.52)	0018.79 (006.02)	LOST
2	0034.48 (002.04)	0063.16 (003.13)	0066.19 (005.01)	0053.47 (004.20)
3	0031.48 (007.90)	0116.47 (012.19)	0083.16 (002.60)	0060.34 (004.72)
4	0040.64 (004.26)	0095.49 (006.50)	0073.59 (002.42)	0024.35 (005.43)
5	0068.40 (019.30)	0054.78 (000.35)	0055.56 (004.99)	0079.06 (001.36)
6	0107.38 (002.06)	0065.99 (007.35)	0070.40 (004.57)	0086.50 (003.58)
7	0079.48 (002.15)	0093.29 (012.85)	0017.50 (000.29)	0044.94 (000.34)
8	0073.95 (005.54)	0092.57 (004.89)	0050.60 (003.37)	0074.54 (006.93)
9	0062.46 (004.00)	0055.78 (004.15)	0063.60 (005.76)	0034.32 (000.69)
10	0056.42 (007.72)	0054.38 (006.59)	0025.26 (001.40)	0047.44 (004.11)
11	0064.50 (009.98)	0055.19 (002.85)	0060.98 (000.67)	0024.85 (003.20)
12	0063.55 (004.48)	0132.49 (001.35)	0061.50 (004.17)	0078.62 (011.38)
13	0033.46 (004.62)	0049.64 (002.57)	0025.24 (003.12)	0065.78 (003.61)
14	0103.82 (001.02)	0042.09 (006.30)	0084.35 (003.85)	0028.37 (002.87)

Table 2.3.6.4. Parameters of remedially treated distribution poles chosen for adjacent soil chemical analysis for fluoride and chromium content (standard deviations in parenthesis for means of 7).

Pole No.	Moisture Content (%)	Diameter (cm)	Creosote Depth (%R)	Pole Depth (m)	Pole Height (m)
1	55	18.4	54.2	1.63	8.17
2	35	19.1	47.1	1.73	8.67
3	45	21.0'	66.6	1.42	7.08
4	50	19.7	50.7	1.42	7.08
5	35	20.4	78.5	1.63	8.17
6	40	18.2	66.1	1.32	6.58
7	45	23.9	62.8	1.42	7.08
8	18	25.8	62.1	1.83	9.17
9	17	20.4	58.9	1.63	8.17
10	17	24.8	64.4	1.93	9.67
11	18	23.6	55.2	1.52	7.58
12	18	18.4	54.2	1.52	7.58
13	17	19.1	73.3	1.52	7.58
14	18	19.1	31.4	1.73	8.67
	Mean	Mean	Mean	Mean	Mean
1-7	43.6 (7.5)	20.1 (1.96)	60.9 (10.9)	1.51 (0.15)	7.55 (0.78)
8-14	17.6 (0.5)	21.6 (3.06)	57.1 (13.0)	1.67 (0.17)	8.34 (0.85)

Table 2.3.6.5. Fitted mean soil concentrations and 95% confidence intervals of fluoride (A) and chromium (B) for the main effects of time after pole treatment, distance of soil sample from pole, and their interaction.

A	Interaction: Time x Distance			Distance	Time	
Time	Distance	Mean F (ug/g)	95% C.I.	Mean F (ug/g)	Time	Mean F (ug/g)
1	6 cm	629.55	(532.19-744.71)	837.15 (772.78-906.87)	1	368.34 (337.31-402.22)
2		759.76	(647.42-891.58)			
3		1033.80	(880.95-1213.18)		2	476.75 (437.47-519.57)
4		991.28	(848.10-1158.64)			
1	25 cm	339.34	(294.42-391.11)	407.08 (379.56-436.59)	3	569.07 (522.17-620.17)
2		435.72	(378.04-502.20)			
3		618.32	(536.46-712.66)		4	420.31 (385.68-458.06)
4		300.97	(261.12-346.89)			
1	> 50 m	234.16	(203.16-269.89)	272.33 (253.41-292.66)		
2		327.34	(284.01-377.28)			
3		288.30	(250.13-332.29)			
4		248.64	(214.43-288.30)			

B	Interaction: Time x Distance			Distance	Time	
Time	Distance	Mean Cr (ug/g)	95% C.I.	Mean Cr (ug/g)	Time	Mean Cr (ug/g)
1	6 cm	212.09	(171.57-262.17)	187.92 (171.74-205.61)	1	116.05 (104.58-128.77)
2		192.67	(162.23-228.83)			
3		201.34	(171.57-236.28)		2	116.40' (105.74-128.12)
4		151.56	(128.38-178.93)			
1	25 cm	123.84	(104.90-146.20)	112.28 (103.65-121.63)	3	111.50' (101.70-122.24)
2		118.98	(101.39-139.63)			
3		144.17	(122.85-169.19)		4	82.43 (075.04-090.56)
4		074.89	(063.82-087.88)			
1	> 50 m	059.38	(050.60-069.69)	55.76 (051.47-060.40)		
2		068.85	(058.67-080.80)			
3		047.80	(040.73-056.09)			
4		049.35	(041.80-058.26)			

Table 2.3.6.6. Fitted mean soil concentrations and 95% confidence intervals of fluoride (A) and chromium (B) for the main effects of time after pole treatment, distance of soil sample from pole, and their interaction.

A	Interaction: Time x Distance			Distance		Time	
Time	Distance	Mean F (ug/g)	95% C.I.	Mean F (ug/g)		Time	Mean F (ug/g)
1	6 cm	618.93	(526.37-727.78)	840.50 (777.43-908.69)		1'	457.60 (411.58-508.77)
2		791.56	(678.58-923.34)				
3		1020.45	(874.80-1190.35)			2'	586.40 (528.48-650.67)
4		998.25	(859.20-1159.80)				
1	25 cm	337.98	(295.01-387.22)	405.86 (379.18-434.41)		3	793.14 (716.23-878.31)
2		433.98	(378.80-497.20)				
3		615.85	(537.54-705.56)			4	547.30 (494.23-606.07)
4		299.76	(261.65-343.44)				

B	Interaction: Time x Distance			Distance		Time	
Time	Distance	Mean Cr (ug/g	95% C.I.	Mean Cr (ug/g)		Time	Mean Cr(ug/g)
1	6 cm	202.35	(163.37-250.64)	194.81 (178.39-212.72)		1'	159.02 (138.80-182.18)
2		214.22	(181.82-252.40)				
3		201.95	(172.09-236.98)			2	159.81 (142.59-179.11)
4		164.19	(139.91-192.67)				
1	25 cm	125.09	(105.95-147.67)	112.73 (104.06-122.12)		3'	170.72 (152.32-191.33)
2		119.22	(101.60-139.91)				
3		144.46	(123.10-169.52)			4	111.05 (099.09-124.46)
4		075.04	(063.94-088.06)				

2.3.6.3. Factors influencing soil concentrations of fluoride and chromium adjacent to remedially treated poles.

In order to determine statistically significant relationships for these data (tables 2.3.6.1 - 2.3.6.3) for the factors of sampling time and distance, an analysis of variance was carried out for each element. Throughout each analysis, the statistical checks detailed in section 2.3.3.5. were made to ensure the validity of the statistical model. In each case, a log transformation was employed to normalise the variance of the data. Analysis of fluoride data was carried out on a reduced number of 309 observations. Twenty five observations over a value of 2000 ug/g, approximately 7.5 % of the original total, were discarded as outliers. Statistical analysis of chromium values employed 314 observations representing approximately 95 % of the original total; high values in excess of 400 ug/g being discarded.

Very highly significant trends were identified for the main or separate effects of time and distance and for their interaction, $P < 0.0005$, 0.0005 and 0.0005 respectively, for both preservative elements (table 2.3.6.5).

The mean fluoride concentration of soil samples 6 cm downslope of poles at 6 and 12 months after treatment was significantly greater than at 1 week. This indicated an increasing loss of fluoride from poles over the first 6 months after sampling, which stabilised over the next 6 months. The soil fluoride concentration at 25 cm from poles 6 months after treatment was significantly greater than that for all other sampling times at this distance which confirmed the increasing loss of preservative fluoride to the soil up to this time. At 12 months after treatment the mean fluoride value at 25 cm from poles was significantly lower than that at 1 and 6 months at this distance and was not significantly different from background levels at any sampling time. Therefore the lateral movement of leached preservative fluoride appeared to be decreasing at this stage. These events were confirmed by the fluoride concentrations in soil for the main effect of time which indicated a rise in fluoride levels up to 6 months (1 and 6 months significantly greater than 1 week) till 12

months, when a significant reduction below the 6 month level took place.

Though the mean chromium concentration of soil samples at 6 cm did not differ significantly at any time after treatment, the mean concentration of this element was substantially lower at 12 months, at this distance. This lower value was reflected in the significant decrease in mean soil chromium concentration 25 cm downslope of poles at this time compared with values for earlier soil samples at this distance. Again, this indication of a decline in leaching loss of chromium from treated poles was confirmed from the mean soil chromium concentrations for the main effect of time which displayed a significant reduction at 12 months after treatment.

2.3.6.4. Pole parameters influencing soil concentrations of fluoride and chromium adjacent to remedially treated poles.

To examine any relationship between remedially treated pole parameters another analysis of variance was carried out using fluoride and chromium values at 6 and 25 cm from poles (table 2.3.6.1 and 2.3.6.2) with pole parameters (table 2.3.6.4) included as co-variates. Background values of both elements (table 2.3.6.3) were excluded as these would not be related to preservative fluoride and chromium in treated poles.

Analysis of fluoride data was carried out on a reduced set of 199 observations. Twenty five outlying values over 2000 ug/g were discarded, representing approximately 11 % of the total. Eleven outlying values over 600 ug/g were discarded during analysis of chromium data leaving 209 observations or 95 % of the original total. Log transformations were carried out as before. Pole moisture content was the only significant co-variate, $P < 0.0005$ and $P = 0.003$ for fluoride and chromium respectively. Higher pole moisture content was associated with an increase in the soil concentrations of both elements.

Very highly significant trends were identified again for the main effects of time and distance for both elements, $P < 0.0005$ and 0.0005 (table 2.3.6.6). The interaction of the main effects was very highly significant for fluoride values, $P < 0.0005$, but less so for chromium values, $P = 0.045$. As expected, examination of the data in table 2.3.6.6 identified no statistical dissimilarities with the data in table 2.3.6.5, which are described in section 2.3.6.3.

2.3.7. Fluoride and chromium losses from remedially treated creosoted pole sections after field exposure.

2.3.7.1. Introduction.

Sawdust samples from 3 groups of aged and creosoted Rentex treated pole sections at 2 field sites, at Tealing in the east of Scotland and Oban in the west of Scotland, were analysed for fluoride and chromium content (section 2.2.6.) to identify the extent of preservative loss in response to field exposure.

2.3.7.2. Results tables.

Table 2.3.7.1 shows the fluoride and chromium concentrations found in 7 remedially treated pole sections, designated TU, which were treated at a field site at Tealing in the east of Scotland and immediately removed for 2 years storage indoors. Table 2.3.7.2 shows these concentrations for 7 pole sections, designated TL, which were subjected to 2 years field exposure at Tealing after remedial treatment. Table 2.3.7.3 displays the concentrations of fluoride and chromium found in 5 pole sections, OL, after field exposure for 4.25 years at Oban in the west of Scotland.

The mean parameters of all sampled pole sections (tables 2.3.7.1 - 2.3.7.3) are shown in table 2.3.7.4.

Table 2.3.7.5 shows the mean fluoride and chromium concentrations combined for each sampling position of each group of pole sections, TU, TL and OL, and the mean values of fluoride and chromium combined for all sampling positions of each group.

Table 2.3.7.1. Mean percentage fluoride and chromium concentrations of wood samples recovered from the groundline (2) and 175 mm above and below the groundline (1, 3) of unexposed remedially treated pole sections, 2 years after treatment: TU (standard deviations* and standard errors** in parenthesis for means of 2 and 6 respectively).

Pole No.	Sampling Position	Mean Fluoride Concentration (% w/w) *	Mean Chromium Concentration (% w/w) *	Mean Fluoride Concentration (% w/w) **	Mean Chromium Concentration (% w/w) **
1	1	1.1634 (0.1395)	0.4563 (0.0407)	0.9684	0.4293
	2	1.0179 (0.0715)	0.4705 (0.0305)	(0.0883)	(0.0239)
	3	0.7239 (0.0944)	0.3612 (0.0203)		
2	1	1.2532 (0.1009)	0.4191 (0.0246)	0.8561	0.3447
	2	0.7406 (0.0708)	0.3426 (0.0218)	(0.1323)	(0.0278)
	3	0.5744 (0.0941)	0.2725 (0.0251)		
3	1	0.9816 (0.0828)	0.2511 (0.0221)	0.8892	0.2461
	2	1.1204 (0.0854)	0.3676 (0.0305)	(0.1077)	(0.0458)
	3	0.5656 (0.0073)	0.1196 (0.0069)		
4	1	0.5260 (0.0108)	0.2319 (0.0205)	0.5335	0.2306
	2	0.5398 (0.0350)	0.2134 (0.0212)	(0.0073)	(0.0099)
	3	0.5349 (0.0076)	0.2463 (0.0317)		
5	1	1.1543 (0.0653)	0.5137 (0.0623)	0.9728	0.4705
	2	1.1575 (0.1305)	0.4374 (0.0411)	(0.1203)	(0.0199)
	3	0.6065 (0.1005)	0.4603 (0.0130)		
6	1	0.7347 (0.1408)	0.2763 (0.0112)	0.6027	0.3073
	2	0.4434 (0.1265)	0.2846 (0.0279)	(0.0652)	(0.0184)
	3	0.6300 (0.0677)	0.3611 (0.0234)		
7	1	0.4006 (0.0283)	0.1328 (0.0075)	0.3978	0.1347
	2	0.4072 (0.0149)	0.1546 (0.0126)	(0.0072)	(0.0080)
	3	0.3856 (0.0079)	0.1166 (0.0154)		

Table 2.3.7.2. Mean percentage fluoride and chromium concentrations of wood samples recovered from the groundline (2) and 175 mm above and below the groundline (1, 3) of remedially treated pole sections after 2 years field exposure: TL (standard deviations* and standard errors** in parenthesis for means of 2 and 6 respectively)

Pole No.	Sampling Position	Mean Fluoride Concentration (% w/w) *	Mean Chromium Concentration (% w/w) *	Mean Fluoride Concentration (% w/w) **	Mean Chromium Concentration (% w/w) **
1	1	0.4495 (0.0202)	0.3149 (0.0296)	0.3646	0.2427
	2	0.4454 (0.0171)	0.2412 (0.0022)	(0.0531)	(0.0267)
	3	0.1989 (0.0369)	0.1720 (0.0077)		
2	1	0.2307 (0.0302)	0.1234 (0.0275)	0.1989	0.1298
	2	0.2142 (0.0033)	0.1605 (0.0079)	(0.0162)	(0.0122)
	3	0.1520 (0.0056)	0.1053 (0.0218)		
3	1	0.2824 (0.0150)	0.2134 (0.0305)	0.2568	0.1822
	2	0.2658 (0.0269)	0.1861 (0.0078)	(0.0127)	(0.0141)
	3	0.2224 (0.0042)	0.1472 (0.0235)		
4	1	0.3422 (0.0120)	0.1111 (0.0014)	0.2989	0.1547
	2	0.2585 (0.0092)	0.1615 (0.0092)	(0.0156)	(0.0167)
	3	0.2959 (0.0023)	0.1914 (0.0411)		
5	1	0.3658 (0.0252)	0.2410 (0.0250)	0.3722	0.1986
	2	0.4854 (0.0769)	0.1637 (0.0111)	(0.0428)	(0.0156)
	3	0.2654 (0.0250)	0.1913 (0.0196)		
6	1	0.3585 (0.0115)	0.2038 (0.0031)	0.3358	0.2007
	2	0.3770 (0.0108)	0.2177 (0.0223)	(0.0208)	(0.0102)
	3	0.2718 (0.0113)	0.1806 (0.0348)		
7	1	0.3656 (0.0370)	0.2169 (0.0285)	0.5975	0.277
	2	0.7277 (0.0708)	0.4304 (0.0330)	(0.0752)	(0.0496)
	3	0.6993 (0.0348)	0.1838 (0.0118)		

Table 2.3.7.3. Mean percentage fluoride and chromium concentrations of wood samples recovered from the groundline (2) and 175 mm above and below the groundline (1, 3) of remedially treated pole sections after 4.25 years field exposure: OL (standard deviations* and standard errors** in parenthesis for means of 2 and 6 respectively).

Pole No.	Sampling Position	Mean Fluoride Concentration (% w/w) *	Mean Chromium Concentration (% w/w) *	Mean Fluoride Concentration (% w/w) **	Mean Chromium Concentration (% w/w) **
1	1	0.9950 (0.1100)	0.4785 (0.0479)	0.8248	0.4115
	2	1.0058 (0.0511)	0.4334 (0.0542)	(0.1139)	(0.0322)
	3	0.4737 (0.0676)	0.3227 (0.0150)		
2	1	0.6226 (0.0756)	0.2323 (0.0211)	0.5567	0.2154
	2	0.4577 (0.0535)	0.1597 (0.0061)	(0.0408)	(0.0190)
	3	0.5897 (0.1048)	0.2541 (0.0237)		
3	1	0.4994 (0.0945)	0.1372 (0.0051)	0.8639	0.3434
	2	1.3494 (0.0749)	0.5319 (0.0870)	(0.1614)	(0.0740)
	3	0.7429 (0.0115)	0.3612 (0.0146)		
4	1	0.6044 (0.0044)	0.3303 (0.0410)	0.5203	0.2627
	2	0.6058 (0.0070)	0.2294 (0.0123)	(0.0538)	(0.0228)
	3	0.3506 (0.0209)	0.2282 (0.0048)		
5	1	1.3181 (0.1044)	0.5725 (0.0417)	0.9694	0.4411
	2	0.9368 (0.1438)	0.4306 (0.0140)	(0.1269)	(0.0470)
	3	0.6531 (0.0779)	0.3203 (0.0226)		

Table 2.3.7.4. Mean pole parameters of field exposed, L, and unexposed, U, remedially treated pole sections recovered from field sites at Tealing, T, and Oban, O (standard deviations in parenthesis for means of 7* and 5**).

Pole Group	Mean Moisture Content (%)	Mean Pole Diameter (cm)	Mean Number of Preservative Injections	Mean Creosote Depth (%R)	Mean Wood Density (g/cm)
TU *	15.00 (00.00)	18.03 (02.46)	80 (12.79)	53.71 (07.77)	0.4931 (0.0896)
TL *	20.36 (03.81)	20.20 (01.86)	84 (12.28)	46.77 (05.35)	0.5184 (0.0676)
OL **	45.40 (14.32)	18.00 (01.37)	Unknown	41.58 (13.28)	0.5060 (0.0622)

Table 2.3.7.5. Mean percentage fluoride and chromium concentrations of wood samples recovered from the groundline (2) and 175 mm above and below the groundline (1, 3) of field exposed (TL, OL) and unexposed (TU) remedially treated pole sections, combined for each sampling position, 1-3, and for all sampling positions of each group of pole sections, TU, TL and OL (standard deviations* and standard errors** in parenthesis for means of 14 and 42 (TU, TL) and 10 and 30 (OL) respectively.

Pole Group	Sampling Position	Mean Fluoride Concentration (% w/w) *	Mean Chromium Concentration (% w/w) *	Mean Fluoride Concentration (% w/w) **	Mean Chromium Concentration (% w/w) **
TU	1	0.8877 (0.0884)	0.3259 (0.0361)	0.7458	0.309
	2	0.7753 (0.0847)	0.3244 (0.0300)	(0.0457)	(0.0190)
	3	0.5744 (0.0297)	0.2768 (0.0333)		
TL	1	0.3421 (0.0184)	0.2035 (0.0185)	0.3464	0.198
	2	0.3963 (0.0464)	0.2230 (0.0250)	(0.0231)	(0.0112)
	3	0.3008 (0.0470)	0.1674 (0.0094)		
OL	1	0.8079 (0.1038)	0.3502 (0.0535)	0.747	0.3348
	2	0.8711 (0.1063)	0.3570 (0.0477)	(0.0560)	(0.0242)
	3	0.5620 (0.0484)	0.2973 (0.0167)		

2.3.7.3. Pole section parameters.

Oneway analysis of variance indicated no significant differences between pole groups TU, TL and OL, for the mean pole parameters of diameter, creosote depth, density or number of injections (for TU and TL). However, mean moisture contents were significantly different between TU and TL, $P = 0.004$, TU and OL, $P < 0.0005$, and TL and OL, $P = 0.002$, ie. TU<TL<OL (table 2.3.7.4).

A count of preservative incisions on the surface of transverse discs cut from each treated pole section (section 2.2.6.4) showed that the mean number and standard deviation of incisions for discs of pole groups TU, TL and OL was 7.00 (1.29), 6.14 (1.68) and 11.75 (0.96) respectively. This indicated that the pole sections erected at Oban had originally received substantially more preservative than the Tealing pole sections.

2.3.7.4. Fluoride and chromium concentrations in field exposed pole sections.

Oneway analysis of variance of the mean fluoride concentrations combined for all sampling positions of each pole section within TU (table 2.3.7.1) indicated that there were significant differences in fluoride concentration between these pole sections, $P < 0.0005$. A similar comparison of chromium concentrations indicated an identical significant difference between these pole sections. The same comparisons between TL pole sections (table 2.3.7.2) indicated an identical significant difference in fluoride concentrations between pole sections and significant differences in mean chromium values, $P = 0.003$. For OL pole sections (table 2.3.7.3) mean fluoride values were significantly different, $P = 0.027$, as were mean chromium values, $P = 0.005$. Though the magnitude of significant differences, between preservative element concentrations of pole sections within each group, decreased as period of field exposure increased, the variability between pole sections of each group indicated that the injection procedure (sections 1.6.2. and 2.1.2.) did not provide equivalent

preservative loadings in treated distribution poles.

The mean fluoride or chromium concentrations from identically numbered sample positions of pole groups TU and OL (table 2.3.7.5) were not significantly different. The same comparisons between TU and TL indicated that mean fluoride or chromium concentrations of TL sample positions 1, 2 and 3 were all significantly lower than those of TU, $P < \text{or} = 0.001$ and $P < \text{or} = 0.015$ respectively. Similarly, mean fluoride or chromium concentrations of TL sample positions 1, 2 and 3 were all significantly lower than those of OL, $P < \text{or} = 0.001$ and $P < \text{or} = 0.013$ respectively. Preservative element concentrations combined for all sample positions of each pole group TU and OL (table 2.3.7.5) were not significantly different. However, the same mean values of fluoride or chromium for TU were significantly greater than for TL, $P < 0.0005$ for both, and the mean concentration of each preservative element for TL was significantly lower than for OL, $P < 0.0005$ for both.

The group of pole sections subjected to 2 years field exposure at the Tealing site (TL) therefore contained significantly less fluoride and chromium, combined with a significantly higher mean moisture content (section 2.3.7.3.) than the group from this site which had been stored indoors for 2 years after treatment (TU). These findings indicated substantial losses of applied preservative, from the field exposed pole sections, probably due to leaching. Though the mean fluoride and chromium concentrations of the OL and TU groups were not significantly different and significantly greater concentrations of both elements were found in pole sections of the former group compared with those of the TL group, this did not indicate that element concentrations in pole sections erected at Oban (OL) had remained stable during 4.25 years field exposure. On the contrary, the OL poles had received approximately 1.68 x the number of preservative injections as were received by the TU poles (section 2.3.7.3.). Therefore based on the fluoride and chromium concentrations in the TU poles, levels of fluoride and chromium in the OL poles had apparently fallen from approximately 1.2519 and 0.5187 % w/w to the present levels of 0.7470 and 0.3348 % w/w (table 2.3.7.5) respectively after 4.25 years field exposure.

2.3.8. Fluoride and chromium concentrations in soils adjacent to remedially treated creosoted pole sections.

2.3.8.1. Soil analysis data.

Table 2.3.8.1, part A, shows the mean fluoride and chromium concentrations in soil samples recovered from the surface of 7 pole sections (designated TL in section 2.3.7.) which were erected, treated and subjected to 2 years field exposure. Background concentrations of fluoride and chromium, > 20 m distant from pole sections, are also given. All samples were recovered as pole sections were uplifted (section 2.2.7.).

Table 2.3.8.1, part B, shows mean fluoride and chromium concentrations in soil samples similarly recovered from sites adjacent to 5 remedially treated pole sections (designated OL in section 2.3.7.) which were erected, treated and subjected to 4.25 years field exposure.

2.3.8.2. Soil concentrations of fluoride and chromium.

Oneway analysis of variance indicated that within background and pole surface soil samples at the Tealing site (tables 2.3.8.1, part A), mean fluoride levels were significantly different, $P = 0.001$ and $P < 0.0005$ respectively. Whereas pole surface soils also displayed significantly different concentrations of chromium, $P < 0.0005$, no significant differences were found between background chromium levels. Mean fluoride or chromium concentrations combined for the 7 pole surface soil samples at the Tealing site were significantly greater than respective mean background concentrations, $P < 0.0005$ for both (table 2.3.8.1, part A).

Table 2.3.8.1. Mean fluoride and chromium concentrations of soil samples adjacent to and > 20 m distant from treated pole sections at A, the Tealing site and B, the Oban site at 2 and 4.25 years after remedial treatment respectively (standard deviations* and standard errors** in parenthesis for means of 2 and 14, A, and 2 and 10, B respectively).

	Sample Distance from Pole	Pole No.	Mean Soil Concentration of: Fluoride (ug/g) *	Chromium (ug/g) *	Fluoride (ug/g) **	Chromium (ug/g) **
A	Pole Surface	1	2418.50 (108.70)	492.23 (12.71)	4473 (907.00)	373.70' (48.10)
		2	6920.00 (270.00)	555.70 (35.50)		
		3	2673.00 (165.00)	167.70 (81.40)		
		4	7609.00 (240.00)	398.10 (57.50)		
		5	9677.00 (839.00)	601.80 (31.50)		
		6	0833.10 (047.60)	169.07 (04.39)		
		7	1176.60 (072.60)	231.00 (10.10)		
	> 20 m	1	232.54 (01.74)	60.62 (11.30)	286.30' (14.60)	58.75 (1.75)
		2	231.33 (07.76)	62.31 (02.85)		
		3	259.23 (06.12)	54.85 (04.70)		
		4	263.30 (31.70)	59.23 (03.57)		
		5	304.08 (07.71)	56.48 (04.95)		
		6	373.80 (41.90)	50.98 (06.03)		
		7	339.80 (09.25)	66.76 (00.56)		
B	Pole Surface	1	0609.17 (014.04)	493.33 (05.02)	876 (131.00)	440.8 (25.80)
		2	0768.49 (011.35)	302.50 (07.28)		
		3	1649.00 (146.00)	513.89 (09.30)		
		4	0711.00 (026.40)	482.54 (05.84)		
		5	0640.60 (059.90)	411.64 (13.34)		
	25 cm	1	549.78 (00.62)	063.22 (08.12)	427.4 (86.60)	68.64 (9.75)
		2	421.70 (31.50)	068.48 (05.02)		
		3	328.28 (02.62	034.38 (02.37)		
		4	482.24 (09.38)	121.77 (03.22)		
		5	354.83 (11.49)	055.36 (06.02)		
	> 20 m	1	317.73 (06.07)	27.05 (05.84)	294.61 (29.55)	51.69 (5.51)
		2	334.57 (11.41)	55.84 (07.68)		
		3	287.27 (02.79)	65.64 (06.48)		
		4	273.12 (01.39)	39.98 (03.86)		
		5	260.38 (06.82)	69.95 (02.50)		

The same statistical comparisons carried out for mean concentrations of both elements in soil samples from the Oban site (table 2.3.8.1, part B) indicated significant differences within pole surface, 25 cm and background samples for both fluoride and chromium, all $P <$ or $= 0.003$. The combined mean concentration of fluoride for pole surface soil samples was significantly greater than that for samples from 25 cm and background, $P = 0.004$ and $P < 0.0005$ respectively. The mean chromium concentration, combined for pole surface samples was also significantly greater than for 25 cm and background samples, $P < 0.0005$ for both. Though the mean fluoride concentration, combined for soil samples at 25 cm from poles was significantly greater than that for background levels, $P < 0.0005$, the combined mean chromium concentrations at these distances were not significantly different (table 2.3.8.1, part B).

These findings clearly indicate that at 2 and 4.25 years after remedial treatment of pole sections, soil concentrations of fluoride and chromium immediately adjacent to the treated timber were greatly in excess of background levels. The evident leaching of preservative constituents was extremely variable from pole to pole. Comparison of table 2.3.8.1, parts A and B clearly shows that chromium concentrations of pole surface soils from each site were not dissimilar despite the greater exposure period of pole sections at the Oban site (table 2.3.8.1, part B). However, the mean fluoride concentration combined for pole surface soil samples was much greater at the Tealing site. This indicated a greater permanence of leached chromium concentrations in soil, which was reflected in the lack of a significant difference between mean chromium soil concentrations combined for 25 cm and > 20 m from pole sections, at the Oban site (table 2.3.8.1, part B), possibly due to restricted lateral movement of leached chromium concentrations.

2.4. DISCUSSION.

2.4.1. Introduction.

As the efficacy of the Rentex preservative for the remedial treatment of creosoted distribution poles of Scots pine is dependant upon the establishment and maintenance of a fungicidal concentration in the uncreosoted groundline areas of the timber, the following discussion of the results of the efficacy studies (section 2.3.) is divided into 2 parts. Section 2.4.2. relates to those studies designed to determine whether adequate fungitoxic fluoride concentrations were established in treated timber, and section 2.4.3. concerns the findings of those studies relevant to the long term maintenance of these concentrations.

2.4.2. Establishment of a preservative effect in remedially treated creosoted distribution poles.

2.4.2.1. Toxicity of Rentex treated wood to selected basidiomycetes and moulds.

2.4.2.1.1. Comparative sensitivity of basidiomycetes and moulds to fluoride concentrations in preservative impregnated Scots pine sapwood.

There were strong indications of a reduction in the virulence of the basidiomycetes ***Neolentinus lepideus*** (BAM 20 and Pole isolate 4) and ***Coriolus versicolor*** (section 2.3.2.3.) and both moulds (section 2.3.2.4.) due to a transfer of preservative from treated sapwood blocks. This was unlikely to have been caused by leaching of preservative constituents from the treated wood blocks as the experimental conditions (section 2.2.1.2.5.) largely precluded this. A more likely explanation is that a gaseous transfer of fluorides from the wood blocks to the atmosphere within the culture vessels took place, and

as this is characteristic of fluoride preservatives in service (Becker and Berghoff, 1963; Becker, 1973, 1976) it is unlikely to have compromised the findings of this study.

The fluoride concentration range in preservative treated Scots pine sapwood specimens required to prevent decay by both strains of the basidiomycete ***N. lepideus*** were very similar at 0.030 - 0.069 % w/w for ***N. lepideus*** (BAM 20) and 0.030 - 0.071 % w/w for ***N. lepideus*** (Pole Isolate 4) (table 2.3.2.11). The equivalent sapwood fluoride concentrations required to provide a protective threshold against decay by ***Coniophora puteana*** and ***C. versicolor*** were approximately twice those which prevented decay by the ***N. lepideus*** strains (table 2.3.2.11) indicating that the former basidiomycetes were more tolerant of fluoride. These data also showed that fluoride concentrations required to prevent sapwood colonisation by the moulds ***Cladosporium resinae*** and ***Trichoderma polysporum***, at 0.59 and 0.29 % w/w respectively, were much higher than those required to prevent decay by basidiomycetes.

These threshold values for ***N. lepideus*** are similar to the 0.016 - 0.064 % w/w equivalent for ***N. lepideus*** quoted by Smith and Cockcroft (1967 b), though they are markedly lower than the 0.2 % w/w equivalent for decay fungi generally quoted by Henningson and Nilsson (1975). However, this latter value is identical to that quoted by Becker (1973) for the fluoride tolerant fungi ***C. puteana*** and ***C. versicolor*** and is close to the values found in the present study for these basidiomycetes. Becker (1973) also named fungi of the species ***Poria*** as more susceptible to fluorides than other basidiomycetes. This confirmed the findings of Richards (1924) who also identified the slightly less susceptible nature of ***N. lepideus*** compared to ***Poria*** species. The threshold value of Henningson and Nilsson (1975) can therefore be considered as an ideal toxic fluoride concentration, rather than a minimum requirement, given that it will be effective against a wide range of basidiomycetes including ***N. lepideus***.

The findings for preservative treated sapwood in this study were therefore in generally good agreement with previous studies in terms of required toxic fluoride concentrations and relative susceptibilities of different basidiomycetes, and as the efficacy of fluoride preservatives against mould fungi is low compared to basidiomycetes (Becker, 1973) the higher toxic thresholds noted for ***C. resinae*** and ***T. polysporum*** in this study were expected.

2.4.2.1.2. Comparative sensitivity of basidiomycetes to fluoride concentrations in preservative impregnated Scots pine heartwood.

The fluoride concentration thresholds required to prevent decay of preservative treated heartwood specimens by ***N. lepideus***, BAM 20 and Pole isolate 4, lay between 0.030 - 0.069 and 0.07 - 0.16 % w/w respectively, whereas for ***C. puteana*** the toxic fluoride concentration lay between 0.08 - 0.16 % w/w (table 2.3.2.11). Therefore the greater tolerance of ***C. puteana*** to fluoride in comparison to both strains of ***N. lepideus*** in sapwood (section 2.4.2.1.1.) was only maintained over the BAM 20 strain in heartwood.

Given that Scots pine heartwood is generally recognised as more resistant to decay than sapwood (Smith and Cockcroft, 1967 b; Wilkinson, 1979: King, 1981; Evans *et al*, 1988) due to the presence in the former of fungitoxic phenolic extractives, it was expected that lower concentrations of fluoride would be required to prevent decay of this wood type in comparison to sapwood. However, the fluoride concentrations required to prevent heartwood decay by ***C. puteana*** and ***L. lepideus*** (BAM 20) were very similar to the equivalent sapwood concentrations for these basidiomycetes (section 2.4.2.1.1.), and the protective fluoride threshold in heartwood preventing decay by the ***N. lepideus***, pole isolate, was double that required to ensure sapwood protection against this basidiomycete. This indicated that the heartwood used in this study was, at least, as susceptible to decay as sapwood and this was confirmed by the similarity of percentage weight losses of untreated wood block *virulence control* specimens of heartwood and sapwood when exposed to identical basidiomycetes (section 2.3.2.2.).

The similar decay susceptibilities of sapwood and heartwood was an unusual finding but may be explained by possible effects of the vacuum impregnation procedure (section 2.2.1.2.2.) on the morphology of the heartwood specimens; specifically the bordered pits linking the vertically orientated tracheids within the wood. The heartwood of Gymnosperms, such as Scots pine, is relatively impermeable to wood preservatives (Wardrop and Davies, 1961; Evans *et al*, 1988) and this is dictated to a large extent by constraints imposed, on liquid movement, by aspiration, or closure, of the bordered pits (Hunt and Garratt, 1953, cited by Wardrop and Davies, 1961). However, comparing the uptakes of identical aqueous concentrations of Rentex between sapwood specimens (tables 2.3.2.2, 2.3.2.4, 2.3.2.6 and 2.3.2.8) and heartwood specimens (tables 2.3.2.3, 2.3.2.5 and 2.3.2.7) indicates that though differences existed, these did not consistently favour either wood type, which suggests that the aforementioned morphological constraints had been removed. It is likely that the pressure imposed within each wood block by the impregnation procedure (section 2.2.1.2.2.) had disrupted the pit membranes in the heartwood specimens thereby removing this morphological difference between heartwood and sapwood.

As the pit apertures are the normal means by which micro-organisms proliferate through wood cells (King and Eggins, 1977; King, 1981) their closure in Scots pine heartwood under normal conditions will hinder the spread of wood rotting fungi and add to the decay resistance of this wood type. It follows that the heartwood specimens used in this study did not present such barriers to the spread of basidiomycetes. As naturally occurring heartwood toxins, which would be retained in these wood blocks, did not appear to restrict the decay process in comparison to sapwood blocks, it appears that the durability of Scots pine heartwood used in this study, with regard to decay by ***C. puteana*** and both strains of ***N. lepideus***, was primarily based on the protection afforded by the presence of aspirated pits.

Given that the method of preservative impregnation probably lowered the decay resistance of the heartwood used in this study, the fluoride concentrations determined as providing a protective threshold against decay by ***C. puteana*** and ***N. lepideus*** strains may

be considered as greater than that which would be required in field poles.

2.4.2.2. Fluoride concentrations within the uncreosoted groundline area of field exposed Rentex treated timber.

Though the majority of fluoride was retained at the preservative injection sites in remedially treated Scots pine pole sections during 20 months of field exposure (section 2.3.3.3.), fluoride was shown to diffuse from these sites throughout the uncreosoted groundline area of the pole sections (section 2.3.3.5.). In the absence of representative findings for wood samples from pole sections recovered from the field 5 months after remedial treatment, due to outlying fluoride values for some of these samples being discarded during the statistical analysis, increasing mean concentrations of diffused fluoride were generally found in the uncreosoted areas of pole sections up to 12 months after treatment (section 2.3.3.5.). All but 1 of these mean fluoride concentrations were in excess of 0.03 % w/w and for almost 80 % of these values the lower 95 % confidence interval was in excess of 0.03 % (table 2.3.3.8). There was no significant difference between the mean fluoride concentration throughout the uncreosoted groundline region of pole sections at 12 months after preservative injection (table 2.3.3.7) and the mean fluoride concentration combined for cores removed from 11 field poles 18 months after identical preservative treatment (table 2.3.5.1).

Though the minimum wood moisture content required for effective diffusion of the majority of preservatives applied by non-pressure methods is considered to be the fibre saturation point, ie. approximately 30 % moisture content (Becker, 1976), fluoride diffusion occurred despite the majority of wood samples analysed being from pole sections and poles having moisture contents well below this moisture content (tables 2.3.3.1 and 2.3.3.2). This indicates that gaseous diffusion of fluorides, which has been demonstrated in previous studies of fluoride preservatives applied to seasoned timber (Buro and Becker, 1956;

Becker and Berghoff, 1963; Becker, 1973, 1976), is likely to have been a major contributor to the migration of fluorides in this study. In addition, greater concentrations of diffused fluoride were associated with wood samples recovered from pole sections of higher moisture content (section 2.3.3.5.) and field poles of higher final moisture content (section 2.3.5.2.1.) which confirms the findings of other workers (Liese and Schubert, 1941; Buro and Becker, 1956; Becker, 1959), where the rate of fluoride diffusion increased with increasing moisture content.

2.4.2.3. The effect of Rentex remedial treatment on the presence of some micro-organisms commonly found in wood cores from the uncreosoted groundline areas of distribution poles.

Due to the small number of ***Neolentinus lepideus*** isolations in distribution poles generally the remedial treatment could not be shown to have had an effect in reducing natural pole populations of this basidiomycete (section 2.3.4.4.2.). For the same reason, a definite treatment effect against strains of the mould ***Trichoderma viride*** could not be established (section 2.3.4.4.6.).

However, remedial treatment caused a significant reduction in the normal bacterial population of distribution poles over the 16 months of the study (section 2.3.4.4.3.). Similarly, with regard to moulds, the micro-organisms most commonly isolated from creosoted distribution poles irrespective of treatment (section 2.3.4.4.1.), the significantly lower mean final presence of these fungi compared to their initial presence in treated poles in conjunction with the lack of a significant difference between these values for control poles, indicates, as for bacteria, that the treatment did have an effect in reducing mould populations (section 2.3.4.4.4.). This treatment effect against moulds was supported by the very highly significant reduction in the normal pole population of strains of ***Cladosporium resinae***, the most commonly isolated mould throughout the study (section 2.3.4.4.6.).

Given the dominating presence of bacteria and moulds especially, in distribution poles (section 2.3.4.4.1.), the aforementioned effects of remedial treatment on pole populations of these micro-organisms were not unexpectedly confirmed by the significantly greater number of wood cores from treated distribution poles from which no micro-organisms were isolated (section 2.3.4.4.5.).

The absence of significant differences in the final mean presence of moulds (section 2.3.4.4.4.) between treated and control poles in no way detracts from the aforementioned indications of treatment effects but demonstrates how natural environmental variations in pole populations of micro-organisms can produce a 'treatment' effect in untreated poles such that any differences between treated and control poles are lost. Alternatively, environmental effects may serve to highlight a treatment effect. For instance, remedial treatment caused a significant reduction in the bacterial population of distribution poles, essentially by removing the normal significant bacterial population imbalance, in favour of 'wet' distribution poles, between 'wet' and 'dry' poles (section 2.3.4.4.3.). Given the more efficient diffusion of fluorides in remedially treated timber of higher moisture content (section 2.4.2.2.) the evident preference of bacteria in this study for timber of higher moisture content (section 2.3.4.4.3.) may have effectively increased the sensitivity of these micro-organisms to fluorides.

2.4.2.4. Conclusions.

The mean fluoride concentrations throughout the uncreosoted areas of the groundline region of distribution poles and pole sections, at 18 and 12 months after Rentex remedial treatment respectively (section 2.4.2.2.), were predominantly greater than those fluoride concentrations in Scots pine sapwood and heartwood determined as providing protection against decay by strains of ***Neolentinus lepideus*** (sections 2.4.2.1.1. and 2.4.2.1.2.), the basidiomycete most commonly associated with internal groundline decay of creosoted

distribution pole stocks in the United Kingdom. This indicated that the groundline area of creosoted distribution poles could reasonably be expected to be protected from the effects of *N. lepideus* for a minimum period of 12-18 months after remedial treatment. However, due to the minor occurrence of this basidiomycete in distribution poles in the field (section 2.4.2.3.) no effect of remedial treatment on pole populations of *N. lepideus* over a 16 month period could be demonstrated.

Remedial treatment did significantly reduce the numbers of bacteria which were isolated from distribution poles (section 2.4.2.3.). However this finding did not indicate preservative efficacy, as bacterial decay of timber in ground contact is a slow process which is not as important as decay through fungal attack, though the extensive porosity of wood caused by bacterial decay may facilitate entry of decay fungi (King, 1981). The real importance of bacteria in wood decay may be in terms of relationships formed between the actinomycete bacteria and decay producing organisms (King and Eggins, 1977) shown in the suppression of *N. lepideus* decay rates in Pine and Lime wood blocks due to the presence of ***Streptomyces xanthochromogenus*** and ***S. bottrophensis*** (Baecker et al, 1981), both bacteria of the actinomycete grouping. In which case, a proven effect of preservative treatment in reducing bacterial numbers alone (section 2.4.2.3.) might be regarded as opposing the natural bio-control achieved by these organisms.

The significant reduction in mould populations generally and ***Cladosporium resinae*** populations in particular in creosoted distribution poles up to 16 months after remedial treatment (section 2.4.2.3.), as for bacteria, did not of itself indicate treatment efficacy as moulds do not generally cause significant strength or weight losses in timber due to their inability to degrade the cellulose and lignin, of wood cells (Butcher, 1966). However, moulds were the micro-organisms most commonly isolated from creosoted distribution poles irrespective of treatment (section 2.4.2.3.), and the fluoride concentration in Scots pine sapwood required to provide a toxic threshold against colonisation by *C. resinae* (section 2.4.2.1.1.) was much greater than that range of fluoride concentrations within

which sapwood and heartwood decay by strains of ***N. lepideus*** was prevented (section 2.4.2.1.1.). Therefore the significant effect of remedial treatment in reducing wood core isolations of ***C. resinae***, strongly indicated that pole populations of ***N. lepideus*** would likewise be significantly reduced up to 16 months after Rentex treatment and hence, the incidence of internal decay in treated creosoted distribution poles would be reduced.

It is clear however, that the fluoride concentrations found throughout the groundline region of field exposed pole sections and poles at 12 and 18 months after remedial treatment respectively (tables 2.3.3.7 and and 2.3.5.1), were much lower than that determined in the laboratory as providing a toxic threshold against colonisation of Scots pine sapwood by ***C. resinae*** (section 2.4.2.1.1.). The increased sensitivity of this organism under field conditions was probably due to the additional effect of environmental stresses, in addition to preservative treatment, which were absent under laboratory conditions. This was not unexpected as there were indications that pole populations of ***C. resinae*** were subject to competition with other inhabitant micro-organisms of field poles (section 2.3.4.4.6.). The microbiological laboratory studies (section 2.4.2.1.) carried out to establish the fluoride concentrations, which were necessary to confer immunity to decay by ***N. lepideus*** in remedially treated uncreosoted timber in the field are likely to be greater than those actually required in these structures. This highlights the inadvisability of relying on laboratory based microbiological studies of preservative treated wood to determine whether the chemical concentrations in preservative treated timber structures in the field are efficacious, in the absence of field based microbiological studies of treated timber.

2.4.3. Long term maintenance of a preservative effect in remedially treated creosoted distribution poles.

2.4.3.1. Fluoride and chromium concentrations within the uncreosoted groundline area of field exposed Rentex treated timber.

Severe reductions in the mean concentrations of fluoride and chromium were found at the sites of preservative injection above the groundline of remedially treated Scots pine pole sections over 20 months of field exposure (section 2.3.3.3.). Similar patterns of decreasing fluoride and chromium concentrations were found when mean values of both elements, combined for all sample positions of treated pole sections, were compared between periods of field exposure (section 2.3.3.4.), indicating that these reductions occurred within the first 5 months of field exposure. These indications were complimented by the progressive falls in the concentrations of both elements found in remedially treated pole sections which were maintained at 2 different field sites for up to 4.25 years after preservative injection (section 2.3.7.4.). As most of the injected fluoride and chromium, in particular, remained at the preservative injection sites (section 2.3.3.3.) these indications of movement of both elements from the groundline area of remedially treated timber suggested that a proportion of both elements was lost by a leaching process.

This appeared to be confirmed by findings which showed that lower concentrations of each element in uncreosoted wood away from the preservative injection sites within the groundline region of field poles, 18 months after remedial treatment (section 2.3.5.), were strongly related to distribution pole height. For instance, lower fluoride concentrations within uncreosoted groundline timber were associated with greater pole height (section 2.3.5.2.1.). Lower chromium concentrations were associated with poles which had butts further below the soil surface (section 2.3.5.2.2.). However, since 1/6 of a distribution poles length is buried to ensure sufficient support for the erect pole, burial depth is therefore directly related to pole height. This pole parameter facilitates the leaching effects of rainfall

as it plays a primary role in increasing the effective pole surface area for rainfall interception and consequent water flow down the pole to the groundline region (Fowlie, 1988).

There were no indications of chromium diffusion from the sites of preservative injection, to other areas of the groundline, in treated pole sections over a period of 20 months after treatment, to match that of fluoride (section 2.3.5.5.). However, and in comparison, respectable chromium concentrations were found in wood cores from distribution poles 18 months after remedial treatment (table 2.3.5.1), when care was taken during core removal to avoid the injection sites (section 2.2.4.). This indicates that a movement of this preservative component to the surrounding timber had taken place which was not noted by the former sampling procedure (section 2.2.2.2.2.). However, as the wood samples recovered from the few mm of wood surrounding the injection sites of treated pole sections (section 2.2.2.2.2.) always included the injection site itself, this was not unexpected and clearly shows that the movement of chromium noted in field poles (table 2.3.5.1.) must have been extremely limited as chromium did not appear outside the immediate injection site area (section 2.3.3.5.). It is unlikely therefore that the movement of chromium was due to an effective diffusive process.

For instance, concentrations of diffusing fluoride within the same field poles were found to increase with higher final timber moisture contents, a very highly significant relationship (section 2.3.5.2.1.), indicating that this pole parameter enhanced fluoride diffusion (section 2.4.2.2.). Though higher chromium concentrations in these poles were likewise associated with higher final timber moisture contents (section 2.3.5.2.2.), suggesting that diffusion played a part in the movement of chromium, this relationship was much less significant than the very highly significant association of higher chromium timber concentrations with lower initial timber moisture contents, ie. moisture content recorded at the time of preservative injection. At the time of preservative injection, as detailed in section 2.1.2., the lower the wood moisture content below the fibre saturation point of around 30 % the greater the impetus for preservative movement into wood by a pressure gradient. Therefore, given that

the majority of initial groundline moisture contents of these field poles were substantially below fibre saturation point (table 2.3.5.2) the limited movement of chromium away from injection sites is likely to have occurred primarily as a result of a pressure impregnation process at the time of injection rather than a diffusion process during the months after injection. A similar statistical relationship between initial wood moisture content and fluoride concentrations was not shown to exist (section 2.3.5.2.1.) undoubtedly because fluoride deposition away from the injection site was primarily dependant upon a diffusive rather than a pressure process, the former masking the latter.

This explanation for chromium movement away from the injection sites is all the more plausible when it is noted that very few of the groundline moisture contents of pole sections (table 2.3.3.1) and field poles (table 2.3.5.2) were at the approximate 30 % fibre saturation point, the minimum moisture content at which effective diffusion of a soluble preservative would be expected to take place (Becker, 1976). As indicated earlier (section 2.4.2.2.), these moisture levels did not restrict fluoride diffusion. The diffusion of soluble chromium (VI), present as sodium dichromate in Rentex, would also be prevented by its rapid reduction to the insoluble cation chromium (III) (Wright and Banks, 1989; Yamamoto and Ruddick, 1992) during the fixation process (Feist and Ellis, 1978). This would result in very limited chromium movement, essentially restricted to the immediate surroundings of the preservative injection sites in Rentex treated pole sections (figures 2.3.3.1 - 2.3.3.3). This pattern of chromium distribution was identical to that described by Graf and Zgraggen (1976), for a wood preservative containing sodium dichromate injected into green wood, ie. unseasoned, where moisture conditions favoured chromium diffusion. As the findings of Nicholas (1972) indicated that the number of active sites available in wood for adsorption of chromium ions is limited, it is apparent that the limited region of wood available for fixation of Rentex chromium, constituting the interior surface of the injection site and its immediate surroundings, would be quickly saturated leaving the excess of unfixed chromium at the injection site prone to leaching along the line of the injection.

In the absence of simultaneous chromium diffusion, the increasing concentrations of diffused fluoride in pole sections up to 12 months after treatment (section 2.4.2.2.) will have remained 'unfixed' and subject to depletion by leaching. Therefore, for these concentrations of fluoride to be maintained it would be necessary for high concentrations of fluoride to be similarly maintained at the preservative injection sites such that an effective concentration gradient was in place to offset continuous leaching losses of 'unfixed' fluoride from the interior regions of the groundline. Given that efficacious concentrations of diffused fluoride found throughout the groundline of remedially treated pole sections up to 12 months after treatment (section 2.4.2.4.), were progressively increasing, it would appear that up to this time fluoride diffusion via a concentration gradient was the dominant process. However, between 12 and 20 months after remedial treatment, fluoride concentrations within the interior of pole sections (table 2.3.3.8.) had fallen below efficacious levels (section 2.4.2.1.) indicating that this situation was reversed in favour of the leaching process, which was not unexpected considering the losses of fluoride from preservative injection sites (section 2.3.3.3.).

2.4.3.2. Fluoride and chromium concentrations in the soil environment adjacent to Rentex treated poles.

The significant increases in concentrations of fluoride and chromium in soils adjacent to remedially treated distribution poles (section 2.3.6.3.) and pole sections (section 2.3.8.2.), which in the former case were encouraged by higher pole moisture contents (section 2.3.6.4.), clearly indicated that remedial preservative treatment of timber in these studies invariably resulted in the leaching of toxic preservative constituents to the surrounding soil environment.

Soil concentrations of fluoride and chromium 6 cm downslope of remedially treated distribution poles were maintained well above background soil concentrations of these elements for up to 1 year after remedial treatment (section 2.3.6.3.), however after the same period of time the soil concentrations of both elements 25 cm downslope of treated distribution poles were reverting to background levels indicating that the greatest loss of preservative constituents occurred within the first year of treatment. Though soil fluoride concentrations 25 cm downslope of remedially treated pole sections were still in excess of background levels after 4.25 years of field exposure (section 2.3.8.2.) these findings must be questioned due to indications that these pole sections had probably received more preservative than other treated timber in these studies (sections 2.3.7.3. and 2.3.7.4.) . However as soil concentrations of both elements in soil from the surface of remedially treated pole sections, which had been subjected to field conditions for 2 and 4.25 years after treatment, were still in excess of background levels, it seems clear that remedially treated timber is likely to lose toxic constituents to the surrounding environment at a lower level over a longer period.

2.4.3.3. Conclusions.

Though diffused fluoride concentrations within uncreosoted groundline areas of remedially treated pole sections, 20 months after treatment, had generally fallen below those concentrations determined in the laboratory as representing a toxic threshold to decay by *N. **lepideus*** (section 2.4.3.1.), it is clear that laboratory estimates of toxic thresholds (section 2.4.2.1.), carried out under conditions divorced from other environmental stresses in the field (section 2.4.2.4.), may have over-estimated the fluoride concentrations required to protect distribution poles. Therefore residual fluoride concentrations (section 2.4.3.1.) after 20 months field exposure may still be effective against this basidiomycete. However, the lack of chromium diffusion (section 2.4.3.1.) and its consequent inability to 'fix' diffused fluoride, ie. fluoride concentrations away from the preservative injection sites, indicates the

transitory nature of toxic fluoride concentrations throughout the groundline area of remedially treated timber. These studies therefore confirm the mobility and impermanence of fluoride concentrations in preservative treated timber which has been found by other workers (Smith and Cockcroft, 1967 b, c; Becker, 1973; Henningson and Nilsson, 1975; Becker, 1976), and question the effectiveness of chromium as a 'fixative' for fluoride in this remedial preservative treatment.

The impermanence of toxic Rentex preservative constituents in remedially treated timbers (section 2.4.3.1.), due to leaching, resulted in soil concentrations of fluoride and chromium in the vicinity of these structures in excess of normal background values (section 2.4.3.2.). The possible harmful environmental impact associated with such contamination around Rentex treated distribution poles is dealt with in chapter 3, which includes a general literature review of environmental studies of wood preservatives (section 3.1.2.).

CHAPTER 3.

ENVIRONMENTAL STUDIES OF REMEDIALLY TREATED DISTRIBUTION POLES USING A PHYSICAL FIELD MODEL.

3.1. INTRODUCTION.

3.1.1. The hazardous nature of wood preservatives.

The Resource Conservation and Recovery Act of the United States, cited by Masters (1991), defines a hazardous substance as one that possesses any of 4 characteristics; reactivity, ignitability, corrosivity or toxicity. Selected wood preservatives fall into all these categories. All are biocides therefore toxic. Reactive wood preservatives include fumigants, creosote oils and waterborne preservatives (as vapour). Copper chrome arsenate preservatives (CCA) are capable of corroding metals with improper use (insufficient 'fixation' periods) and oil based or organic solvent preservatives are easily ignited.

3.1.2. Wood preservatives and the environment: the requirement for impact assessment.

In the light of the number of potential hazards represented by these products (section 3.1.1.), the necessity for environmental assessments of established and more recently adopted wood preservatives is apparent. However as recently as 1985, Hedley and Butcher published a 'Protocol for Evaluating and Approving New Wood Preservatives' which ignored the environmental consequences of preservative use and defined 'hazard' only as the extent to which preservative treated structures were at risk from decay organisms. Willeitner (1973) accepted the hazardous nature of wood preservatives as a necessary property for their successful use and indicated two main areas of environmental concern; wood treatment processes at treatment sites and impregnated timber in service, the former representing the greater risk of environmental contamination. Indeed, the guidelines of the Inter-departmental Committee on the Re-development of Contaminated Land, ICRCL (Anon, 1987), identified soil at wood preservation plants in the United Kingdom as likely to

contain high levels of hazardous contaminants.

The extent to which the use or mis-use of wood preservatives at sites of large scale treatment can affect the local environment was highlighted by Grant and Dobbs (1977) from studies carried out at CCA preservative treatment plants in the United Kingdom. These workers found levels of copper, chromium and arsenic in the top 5 cm of soil as high as 82,000, 45,000 and 73,000 ug/g respectively, which were vastly in excess of normal background levels and were sufficient to completely inhibit the growth of dwarf french bean, carrot, tomato, perennial ryegrass and copper tolerant bentgrass. Similarly, Bergholm (1990) described soils at a number of Swedish CCA wood preservation plants as highly polluted with arsenic, chromium and copper due to direct spillage of preservative solutions and wastes or leaching from treated wood in storage. Occasional soil concentrations of arsenic as high as 20,000 ug/g were found, and topsoils from all the treatment plants examined, contained arsenic levels sufficient to affect or inhibit the growth of ryegrass. More widespread effects of preservative mis-use were identified around a sawmill in the north-east of Scotland (M^{C}Neil, 1989) where river spillages, amounting to 12.1 m^3, of a preservative containing pentachlorophenol, bis (tri-butyl-tin) oxide and dieldrin, resulted in the complete eradication of inverterbrate populations immediately downstream of the site. Continued sub-soil leaching of preservative constituents prevented re-colonisation by all but the most tolerant of species up to 4 years after these events.

As opposed to the evident environmental problems associated with preservative treatment sites, the findings of Degroot *et al* (1979) for CCA and ammoniacal chrome arsenate preservatives (ACA) and Arsenault (1975) for CCA preservatives, would seem to support the view of Willeitner (1973), that "pollution ..... is almost negligible for impregnated timber in use". These workers examined distribution gradients in field soils adjacent to pressure treated wooden stakes and found little evidence of substantial lateral transfer of arsenic or chromium. Degroot *et al* (1979) felt that concentrations leached would not contribute significantly to arsenic and chromium concentrations in surrounding

soils and water. Grant and Dobbs (1977) considered such findings for CCA preservatives unsurprising given their resistance to leaching, presumably due to effective fixation in wood (see section 2.1.5.).

Studies carried out by Levi *et al* (1974), indicated that no uptake of copper, chromium or arsenic was evident in grape vines growing in close proximity to CCA-pressure treated stakes. Similarly no foliar damage was found when the rose cultivar Helsingor was grown adjacent to trellises treated with the CCA preservatives Boliden K33 and Tanalith (Qvarnstrom, 1978 b). However, root development of tulips was inhibited when plants were grown in shallow wooden boxes pressure treated with Boliden K33 (Qvarnstrom, 1982) and the same preservative virtually stopped root development of gladioli bulbs, though aerial plant parts seemed unaffected (Qvarnstrom, 1978 a). Qvarnstrom (1978 a, b, 1982) described various severe phytotoxic effects associated with wood pressure treated with the organic solvent preservative Hylosan PT (containing tri-butyl-tin oxide and benzalkyl-trimethyl-ammonium chloride), and the preservative oil creosote, as well as brush applied organic solvent preservatives such as copper, zinc and iron naphthenates. Similarly, Franco and Baonzo (1989) identified severe foliar damage in toadskin melon and long English cucumber grown on wooden plant supports which had been pressure treated with 3 un-named organic preservatives.

Therefore, though the potential polluting effects of an individual treated timber structure obviously cannot approach that at large scale preservative treatment sites, and may be minimal for some preservative treatments such as CCA, possible environmental effects cannot be ignored, particularly when these effects will be multiplied by the number of structures in service, which may be exposed to a variety of environmental conditions. Degroot *et al* (1979), citing an unpublished report of the International Research Group on Wood Preservation (1969), stated that adequate monitoring of potential changes in soils around experimental test units was minimal, indicating that up to this time, environmental considerations, with regard to wood preservatives, were largely ignored. The need to rectify

this lack of knowledge was highlighted by the guidelines of the ICRCL (Anon, 1987) which identified a wide range of chemicals and compounds, commonly used in wood preservatives, as hazardous soil contaminants. That environmental studies should be considered a particular priority for remedial in-situ timber treatments, such as Rentex groundline treatment of distribution poles, is supported by Willeitner (1973). While maintaining that pollution problems with regard to preservatives was essentially a concern over the treatment process, he suggested that preservative pollution due to these processes could be quantified as contamination per unit of wood treated. This scheme, which was accepted in a recent United Nations Environmental Programme (UNEP) report on the environmental aspects of wood preservation, (Anon, 1994), indicated that environmental contamination was generally greater for small scale treatments at many locations, such as remedial in-situ treatments, as these were subject to less control and containment.

3.1.3. Field measurements indicating a requirement for environmental assessment of Rentex remedial treatment.

The indications of fluoride and chromium loss from Rentex treated timber, presented in sections 2.3.3. and 2.3.7., show that leaching of these preservative constituents occurred during field exposure. This resulted in persistent soil concentrations of fluoride and chromium, adjacent to field poles (section 2.3.6., table 2.3.6.5., parts A and B) and pole sections (section 2.3.8., table 2.3.8.1, parts A and B), significantly greater than normal background values.

All of the mean total chromium concentrations in the primarily agricultural soils adjacent to Rentex treated field poles were in excess of the United Kingdom threshold 'trigger' value of 70 ug/g in soil for cropping (Anon, 1987), at which level the soil is considered contaminated and liable for cleaning-up procedures. These procedures may include removal to another site or mixing with uncontaminated soil. In addition, the

majority of chromium values in soils associated with Rentex treated timber also exceeded 80 ug/g, the recommended upper permitted concentration of chromium in Scottish soils for sewage sludge amendment (Williams, 1988, cited by M^{c}Grath and Smith, 1990).

Irrespective of the questions raised regarding the long term efficacy of the remedial treatment (section 2.4.3.), these findings clearly presented a case for further study to examine any environmental effects associated with the presence of leached chromium and fluoride preservative constituents in soil. Though an analysis of vegetation adjacent to treated timbers in the field would have provided data relevant to any such environmental effects, the activities of local grazing animals prevented the collection of representative plant material (see section 3.1.7.1.).

3.1.4. Environmental impact assessment of Rentex remedial treatment.

In order to provide meaningful assessments of the environmental impact of any chemical within the environment, a logical 3-stage hazard evaluation approach has been advocated by Bro-Rasmussen (1988), which does not differ significantly from that proposed by UNEP for wood preservatives (Anon, 1994). Initially the chemicals' hazardous properties such as toxicity, persistence and environmental mobility must be identified. The second stage is an assessment of the potential for these properties to be translated into adverse effects on individual organisms or natural systems deemed to be at risk due to the specific environmental exposure of the chemical. Then an assessment is required to determine whether or not this potential, if it exists, is realised by virtue of the chemicals exposure and actual measured effects on the 'receptors' or 'indicators' identified at stage 2.

Combining the first 2 stages of this system of hazard evaluation for Rentex, a survey of the literature was carried out to examine environmental effects of fluoride and chromium (sections 3.1.5. and 3.1.6.) with regard to the environmental contamination associated with

the preservative treatment method. As this contamination consisted of elevated concentrations of fluoride and chromium in soils adjacent to remedially treated timber (section 3.1.3.), the phytotoxic effects of both elements were considered particularly important, though more general environmental effects due to possible soil leaching and bioaccumulation from contaminated vegetation were also considered.

3.1.5. Fluoride toxicity, persistence and mobility in the environment with reference to Rentex remedial treatment.

3.1.5.1. Some phytotoxic effects of fluoride.

3.1.5.1.1. Atmospheric fluoride.

Many studies have been carried out to assess phytotoxic effects of atmospheric fluoride (Gilbert, 1975; Mandl *et al*, 1975; Maclean *et al*, 1982; Doley, 1984; Maclean *et al*, 1984; Murray, 1984; Doley, 1986) which penetrates the leaves mainly through the stomata and accumulates in the chloroplasts (Chang and Thompson, 1966). These studies have a direct bearing on possible phytotoxic effects of elevated soil fluoride concentrations around remedially treated distribution poles, as regardless of its path of entry to the plant, fluoride distribution in leaf tissue and its physiological effects appear to be similar (M^{c}Laughlin and Barnes, 1975).

The foliage of certain grapevine (***Vitis vinifera L.***) varieties may accumulate as much fluoride as 1 mg/g with no visible signs of injury (Doley, 1984). However, foliar fluoride concentrations as low as 27 ug/g in this cultivar have induced toxic effects such as foliar necrosis, reduced leaf chlorophyll-a, reduced total chlorophyll content and reduced mature leaf size (Murray, 1984; Doley, 1986), while in some south-east American trees, inhibition of apparent photosynthesis and stimulation of dark respiration was encountered at foliar

fluoride concentrations of < 10 ug/g (M^{c}Laughlin and Barnes, 1975). M^{c}Laughlin and Barnes (1975) found softwoods generally more sensitive than hardwoods. This greater sensitivity of softwoods was confirmed by Gilbert (1975) in a study of particularly severe phytotoxic effects in a variety of vegetation types around a number of Norwegian aluminium smelters, where aluminium fluoride is used to lower the melting point and increase the conductivity of electrolytes during the smelting process (Pelham, 1986).

As well as differences in the severity of phytotoxic effects due to variable susceptibilities of plant species and varieties, phytotoxic effects of fluoride are enhanced by foliar accumulation in conditions of darkness or by low atmospheric concentrations of sulphur dioxide (Mandl *et al*, 1975) and though Maclean *et al* (1984) found that fluoride accumulation in wheat (***Triticum aestivum L.***) and a sorghum hybrid (Northrup King 222A) were related to the mean concentration of atmospheric fluoride over the entire exposure period, yield reductions were most closely related to atmospheric fluoride concentrations at exposures coinciding with seed head emergence.

The typical effect of foliar fluoride accumulation appears to be inhibition of photosynthesis (M^{c}Laughlin and Barnes, 1975; Parry *et al*, 1984) within a timescale which precludes an effect due to a reduction in leaf chlorophyll concentration as a result of possible inhibition of chlorophyll synthesis or enhanced chlorophyll degradation (Murray, 1984). It is more likely that inhibition is caused by effects of fluoride on the enzymes associated with CO_2 fixation. Parry *et al* (1984) for instance, showed that fluoride is a potent inhibitor of both reactions of ribulose-P_2 carboxylase/oxygenase in vitro; CO_2 fixation involving the carboxylation of ribulose-P_2 to 2 molecules of 3-phosphoglycerate, and the oxygenation of ribulose-P_2 to 2-phosphoglycollate and 3-phosphoglycerate in the initial reaction of photorespiration.

3.1.5.1.2. Soil fluoride.

Fluoride is relatively immobile in soil (Larsen and Widdowson, 1971; Gilpin and Johnson, 1980) due to complexation with aluminium (Omueti and Jones, 1980, Mengel and Kirkby, 1982, Farrah *et al*, 1985; Peek and Volk, 1985) and iron compounds (Farrah *et al*, 1985; Peek and Volk,1985) and precipitation as calcium fluoride (Mengel and Kirkby, 1982; Farrah *et al*, 1985). In addition, compared to other anions fluoride is sparingly taken up by plant roots even when in available soluble form (Venkateswarlu *et al*, 1965). These effects of poor availability in the soil and low plant uptake potential account for the normally low levels of fluoride in plants, occurring in the range of 2 - 20 ug/g of plant dry weight (Mengel and Kirkby, 1982). However, modest additions of fluoride as superphosphate fertiliser (Larsen and Widdowson, 1971) and sodium fluoride (Singh *et al*, 1979 b) have been shown to markedly increase soluble fluoride levels in soil irrespective of pH, and as fluoride uptake by plants is directly related to the amount of soluble fluoride present in the growing medium (Leone *et al*, 1948; Hara *et al*, 1977; Singh *et al*, 1979 b) phytotoxic effects may occur as a consequence of the elevated levels of total soil fluoride adjacent to Rentex treated timber.

Singh *et al* (1979 a), for instance, found significant reductions in the grain and dry matter yield of rice plants (***Oryza sativa***) when grown in two alkaline saline soils, as a consequence of sodium fluoride addition. Fluoride additions of 0, 25, 50, 100 and 200 ug/g dry weight of soil resulted in progressively greater concentrations of fluoride in mature rice straw. Fluoride concentrations rose from a level of of approximately 20 ug/g for plants in both control soils, to 60 and 35 ug/g for plants in soils of higher and lower salinity respectively, after fluoride additions to soil of 200 ug/g. Respective grain yield reductions, compared to controls, amounted to approximately 18 and 15 %, and for plants with the highest level of accumulated fluoride, dry matter straw yield was reduced by approximately 13 %. Significant reductions were recorded in the yield of wheat (***Triticum aestivum L.***), which was sown in the soils of lower salinity after the rice plants were harvested, due to

fluoride accumulation which increased linearly with increasing water extractable fluoride brought about by increasing soil additions (Singh *et al*, 1979 b). These workers (Singh *et al*, 1979 b) identified a fluoride content of 31.8 ug/g in mature wheat straw, brought about by soil additions of fluoride of as little as 25 ug/g, as the critical value with respect to significant depression of grain yield, or about 50 % of that critical foliar concentration for yield depression in lucerne (***Medicago sativa***) (Hansen *et al*, 1958).

The greater fluoride uptake found by Singh *et al* (1979 a) for rice plants grown in the more saline soil was ascribed by these workers to the dominant presence in this soil of soluble sodium fluoride as opposed to the dominance of much more insoluble calcium fluoride in the soil of lower salinity, as increased plant uptake of sodium and reduced uptake of calcium, which was enhanced by increasing fluoride additions to both soils, was more pronounced for the soil of higher salinity. The importance of calcium in protecting plants from potentially toxic levels of soluble fluoride was shown by Hansen *et al* (1958) who found that a fluoride application of 800 ug/g to a calcareous soil resulted in only a slight increase in foliar fluoride concentration, whereas in a non-calcareous soil an addition of 200 ug/g doubled the foliar fluoride concentration. However, given sufficient soil concentrations of calcium fluoride, significant plant uptake of fluoride can occur.

Wright *et al* (1978), for instance, found that the fluoride content of indigenous vegetation in the vicinity of a fluorspar (calcium fluoride) tailings dam was clearly related to elevated soil fluoride levels. Mean total soil fluoride concentrations of 174,200, 15,600 and 7,050 ug/g for samples from the dam surface, dam wall and adjacent fields respectively, resulted in plant fluoride concentration ranges of 2040 - 3415, 145 - 378 and 37 - 59 ug/g respectively, depending on the plant species sampled. No phytotoxic effects were recorded by these workers.

3.1.5.2. Bioaccumulation and toxicity of fluoride in terrestrial and aquatic environments.

The findings of Wright *et al* (1978) indicate that fluoride levels in the tissues of long tailed field mice (***Apodemus sylvaticus L.***) and field voles (***Microtus agrestis L.***) displayed a strong relationship with the fluoride content of soils and vegetation within the domain of a fluorspar tailings dam (see section 3.1.5.1.2.). Femur fluoride concentrations, which provide a significant indication of whole body fluoride retention (Wright and Thompson, 1978), measured for individuals collected from the dam surface/wall, adjacent fields and an uncontaminated field site were 4387, 1077 and 189 ug/g respectively for ***A. sylvaticus***, and 2195, 379 and 117 ug/g respectively for ***M. agrestis***. These differences in fluoride accumulation were probably due to differences in the animals' feeding habits. For instance, ***M. agrestis*** is solely herbivorous whereas ***A. sylvaticus*** is omnivorous, its diet including snails and earthworms, (Corbet and Southern, 1977) and a close relationship exists between the fluoride concentration in soil, vegetation and the tissues of arthropods and earthworms (Andrews *et al*, 1982). Given that the little owl (***Athene noctua***) and the tawny owl (***Strix aluco***) accumulate more fluoride than other owls probably due to the consumption of earthworms in addition to their principal diet of small mammals (Seel and Thompson, 1984), arthropods and earthworms may represent more available forms of accumulated fluoride and hence the greater levels of fluoride found in ***A. sylvaticus*** by Wright *et al* (1978). Similarly, high femur fluoride concentrations of 2693 and 2051 ug/g in the merlin (***Falco columbarius***) and sparrowhawk (***Accipiter nisus***) which feed predominantly on small birds, compared with a mean fluoride concentration of 726.6 ug/g in the buzzard (***Buteo buteo***) which takes mostly mammals (Seel and Thompson, 1984) suggests that small birds may be greater accumulators of fluoride than small mammals.

These studies clearly indicate that elevated concentrations of fluoride in soil and vegetation can progress through terrestrial ecosystems. However, Wright *et al* (*1978*) identified no toxic effects of fluoride accumulation in small mammals exposed to

concentrations of fluoride in soil and vegetation (see section 3.1.5.1.2.), greatly in excess of any possible due to Rentex treatment, extending over an area including a dam surface of 7 hectares. Therefore, the possibility of any toxic effects on animal life caused by the magnitude of fluoride soil concentrations localised at remedially treated field poles, via persistent exposure to contaminated soil, vegetation or windblown soil particles is remote.

However, as fluoride is predominantly associated with clay size minerals in soil (Omueti and Jones, 1980; Peek and Volk, 1985) with maximum adsorption of soluble soil fluoride occurring at around pH 6 (Larsen and Widdowson, 1971; Omueti and Jones, 1980; Gupta *et al*, 1982), leaching of fluoride may occur in soils of lower pH and lower clay content (Omueti and Jones, 1980) or at higher pH in soils low in amorphous aluminium (Peek and Volk, 1985). Therefore, the entry of fluoride into nearby aquatic ecosystems via groundwater contamination or fluoride movement in surface run-off waters may be a possibility.

The natural accumulation of fluoride in the tissues of fish, molluscs and crustaceans, which occurs in uncontaminated marine environments (Ke *et al*, 1970; Wright and Davison, 1975), where seawater fluoride concentrations typically range from 0.9 - 1.5 ug/cm^3 (Wright and Davison, 1975; Pankhurst *et al*, 1980), is enhanced by increased ambient fluoride concentrations and exposure times (Hemens *et al*, 1975; Wright and Davison, 1975). Sensitive marine organisms such as the brine shrimp (***Artemia salina***) and the brown mussel (***Perna perna***) displayed toxic symptoms when exposed to seawater fluoride concentrations, in the laboratory, as low as 5 ug/cm^3 for 12 days and < 7 ug/cm^3 for 5 days respectively (Pankhurst *et al*, 1980; Hemens and Warwick, 1971). However, these relatively low toxic concentrations of fluoride do not realistically represent sustained seawater concentrations even in the immediate vicinity of fluoride loaded effluent outfalls due to dispersal and dilution effects and the rapid formation of insoluble fluorides (Wright and Davison, 1975; Pankhurst *et al*, 1980). Therefore the possibility of even minute effects on the marine environment due to fluoride leached from the soil surrounding remedially treated

distribution poles must be regarded as minimal.

However, the entry of fluoride leached from the soil into the marine environment will be largely mediated by its concentration in freshwater river systems. These waters are frequently closely associated with distribution pole lines and may subject leached soil contaminants to less dispersal and dilution than marine waters, therefore localised effects of leached preservative fluoride in freshwater systems cannot be discounted. Freshwater fluoride concentrations as low as 2.7 - 4.7 ug/cm^3, for instance, have been shown to cause 50 % mortality in rainbow trout (***Salmo gairdnerii***) after 21 days exposure (Neuhold and Sigler, 1960), though Herbert and Shurben (1964) recorded no mortalities in the same species when exposed to fluoride concentrations as high as 75 ug/cm^3 for a similar period of time. This disparity in findings was explained by Wright (1977) in terms of the calcium content of the water used by these workers, 4.5 and 18 ug of Ca per cm^3 respectively. Calcium reduces fluoride toxicity by the formation of insoluble calcium/fluoride complexes, in the same way as complexation with fluoride mitigates the toxic effect of aluminium in aquatic ecosystems (Plankey and Patterson, 1986). However, 50 % mortality of brown trout fry (***Salmo trutta***) occurred when exposed for 15 and 75 hours to tapwater with fluoride concentrations of 50 and 20 ug/cm^3 respectively and a calcium content of 29 ug/cm^3 (Wright, 1977). Comparing these findings with those of Herbert and Shurben (1964), for yearling rainbow trout, indicates that decreasing fluoride toxicity by increasing fluoride complexation with calcium failed to overcome the greater sensitivity of brown trout fry, and strongly suggests that, at lower freshwater calcium concentrations, fish fry may be severely affected by freshwater fluoride at concentrations and exposure times even lower than those recorded by Neuhold and Sigler (1960) for rainbow trout.

3.1.6. Chromium toxicity, persistence and mobilty in the environment with reference to Rentex remedial treatment.

3.1.6.1. Chromium behaviour in soil.

The speciation of heavy metals in soil is of prime importance with regard to their mobility and bioavailability (Camerlynck and Kiekens, 1982). The main chromium species' found in soil possess contrasting chemical properties and hence, studies of chromium mobility, bioavailability and toxicity in the environment typically indicate contrasting potential for harmful effects.

Chromium exists in 2 main oxidation states in soil, chromium (VI), that form found in the preservative Rentex (section 1.6.2.), and chromium (III) (Cary, 1982; Bartlett and James, 1988; M^{c}Grath and Smith, 1990). Chromate is in pH dependant equilibrium with other forms of anionic chromium (VI) such as $HCrO_4^-$ and dichromate (Cr_2O_7) with CrO_4^{2-} the predominant form at pH > 6 (Anon, 1981; Bartlett and James, 1988; M^{c}Grath and Smith, 1990) and $HCrO_4^-$ predominating in more acidic environments (Calder, 1988). Chromium (VI) is the more mobile species but it is strongly oxidising and in the presence of soil organic matter is reduced to Cr (III) (Bartlett and Kimble, 1976 b; Cary *et al*, 1977 b; Grove and Ellis, 1980; Bloomfield and Pruden, 1980; Bartlett and James, 1988) which is found as a cation under the acid to neutral conditions (Bartlett and James, 1988; Calder, 1988) which prevail in the soils of the United Kingdom (Alloway, 1990). Chromium (III) is recognised as the more stable form of chromium in most arable soils (Bartlett and Kimble, 1976 a, b; Cary *et al*, 1977 b; Bartlett and James, 1988).

The mobility of Cr (VI) in the soil solution is governed by the extent of its reduction and adsorption, both of which characteristically increase as pH decreases (Bartlett and Kimble, 1976 b; Cary *et al*, 1977 b; Bloomfield and Pruden, 1980; M^{c}Grath, 1982; James and Bartlett, 1983 c), the former due to a requirement for H^+ ions in the reduction process

(Grove and Ellis, 1980) and the latter due to the increase in positive charge of the adsorbing soil medium (Bartlett and Kimble, 1976 b). Adsorbates of Cr (VI) in soils such as commonly found iron, aluminium and manganese oxides and hydroxides (Bartlett and Kimble, 1976 b; James and Bartlett, 1983 c; Calder, 1988; Bartlett and James, 1988) possess a high zero point of charge pH in the range of 6.7 - 8.5 (Stumm and Morgan, 1981) therefore their predominant surface charge above this pH is negative and little or no adsorption of Cr (VI) anions occurs (Bartlett and Kimble, 1976 b; James and Bartlett, 1983 c).

Bartlett and Kimble (1976 b) demonstrated the importance of organic matter and soil pH to Cr (VI) reduction by adding Cr (VI), as potassium dichromate, to acidified (pH 2.9 - 3.5) soil suspensions practically free of organic matter and to near neutral suspensions of the same soil type with added cow manure. Reduction did not occur over a 48 hour period in either treatment, and by using orthophosphate as a competitive inhibitor of Cr (VI) adsorption all the Cr (VI) remained in solution. Combining the 2 treatments resulted in reduction of 99.5 % of added Cr (VI) within 24 hours. Bloomfield and Pruden (1980) used field soil cores of equivalent organic matter content and different pH to study the sequestering of 14.71 mg of Cr (VI), added as sodium dichromate, into reduced and adsorbed forms over a 3 week period. In soil of pH 6.62, reduced and adsorbed Cr (VI) amounted to approximately 7 and 11 % respectively of the total added whereas in soil of pH 4.20 these forms of chromium made up approximately 47 and 32 % of the total. Labile Cr (VI) which was removed unchanged by leaching with water at the end of the experiment made up approximately 82 and 21 % of added Cr (VI) in the near neutral and acid soil respectively. These findings indicate that the removal of Cr (VI) from the labile pool in field soils is not a fast process, even given ideal conditions of acidity and sufficient organic matter.

It is not clear whether adsorbed Cr (VI) represents a renewable source of labile Cr (VI). James and Bartlett (1983 c) presented evidence indicating that Cr (VI) was partially

protected from reduction when adsorbed in certain limed subsoils containing high levels of organic matter and amorphous organically complexed aluminium and iron. However these workers also found that no protection was afforded to adsorbed Cr (VI) in the same unlimed soils and in other limed or unlimed subsoils. Indeed, the findings for other limed soils indicated that adsorbed Cr (VI) was preferentially reduced compared to soluble forms.

The mobility of Cr (III) in soils is restricted above pH 4, and above ph 5.5 complete precipitation occurs due to the low solubility of Cr (III) solid phases Cr_2O_3 and $Cr(OH)_3$ (James and Bartlett, 1983 a; Bartlett and James, 1988; Calder, 1988; M^{C}Grath and Smith, 1990). However, complexing Cr (III) with soluble organic acids such as citric and fulvic acid, and soil extracts of water soluble organic matter can improve solubility and maintain Cr (III) in the soil solution up to pH > 7.5 (Bartlett and Kimble, 1976 a; James and Bartlett, 1983 a, b), though such complexes have not been documented under field conditions. Therefore within the the typical range of soil pH found in the United Kingdom, which Alloway (1990) gives as 4 - 8, Cr (III) will be relatively immobile. In addition, according to Calder (1988), clay mineral adsorbates of Cr (III) commonly have high cation exchange capacities and a low zero point of charge pH in the range of 2 - 2.5 and hence Cr^{3+}, the dominant Cr (III) species below pH 4, is also relatively immobile due to adsorption. Wentink and Etzel (1972) considered Cr (III) as so readily immobilised by soils having moderate to high ion exchange capacities that they suggested Cr (III) be removed from electroplating effluents by percolation through beds of soil.

Bartlett and James (1979) reported that rapid oxidation of a portion of Cr (III) salts or hydroxides added to almost any soil with a pH above 5, took place readily, provided the soil sample was moist and fresh from the field. The amount of Cr (III) oxidised to Cr (VI) was proportional to the manganese reduced (and exchangeable). Complexation with citrate appeared to facilitate the oxidation of older precipitates of Cr (III) by increasing their solubility and mobility (James and Bartlett, 1983 b). However, unlike the intimately mixed moist soil samples used by these workers, field soils in-situ are characteristically

heterogenous, manganese oxide surfaces will be discontinuous, as will soil water films at field moisture capacity and below, resulting in restricted movement of even diffusible ion species which Cr (III) is not. Bartlett and James (1988) concluded that, due to these kinetic problems, the oxidation of Cr (III) in the field would be slow at best and given the opportunities for its reduction, accumulated Cr (VI) from Cr (III) sources may rarely be measureable. By implication, the introduction and maintenance of large concentrations of Cr (VI) in the soil solution of field soils via oxidation of Cr (III) is unlikely to occur.

As the chromium content of Rentex consists of sodium dichromate (section 1.6.2.), a Cr (VI) species, it is likely that the leaching of this soluble species was responsible for a significant portion of the increase in chromium soil concentrations adjacent to remedially treated timber (sections 2.3.6. and 2.3.8.). Given the mobility of this species, increased Cr (VI) concentrations in the soil solution will be favoured. However, with the majority of field soils in the United Kingdom having adequate organic matter and tending towards acidity (Mengel and Kirkby, 1982; Alloway, 1990), the ultimate fate of highly mobile Cr (VI) will be reduction to chemically inert Cr (III). Though the rate of Cr (VI) adsorption and reduction may be prolonged, long term maintenance of Cr (VI) concentrations will largely depend on the continued release of this species from the preservative source, a finite resource, itself subject to adsorption and reduction through reaction within the timber (section 2.4.3.1.).

Chromium (VI) is therefore likely to be a transient species in the soil around Rentex treated timber and any long term environmental effects of chromium soil contamination will most likely arise as a consequence of Cr (III) which, in the absence of oxidation to Cr (VI) or mobilisation via complexation with soluble organic acids, will remain essentially immobile in most soils. Therefore, the literature review of the environmental effects of chromium, with regard to the remedial treatment under study, includes both chromium species (sections 3.1.6.2. and 3.1.6.3.).

3.1.6.2. Toxicity and bioaccumulation of chromium in the terrestrial environment.

3.1.6.2.1. Some phytotoxic effects of chromium.

Both chromium species are toxic to plants (Breeze, 1973; Anon, 1976; Skeffington *et al*, 1976; M^{c}Grath, 1982) and though Cr (VI) is generally regarded as more phytotoxic than Cr (III) (Cary, 1982) other studies have indicated that both forms of chromium are equally damaging when in soluble form at the roots (Breeze, 1973; Anon, 1976; M^{c}Grath, 1982). The phytotoxicity of both species will be largely determined by those soil conditions which favour increased concentrations of chromium in the soil solution (section 3.1.6.1.).

This was amply demonstrated by Breeze (1973) who found that a chromium addition to compost of 5000 ug/g as Cr (III) was required to induce a lethal effect in perennial ryegrass (***L. perenne***) equivalent to a compost amendment of 500 ug/g as Cr (VI). This ten-fold increase in compost additions of Cr (III), compared to Cr (VI), required for a comparable toxic effect, was due to the rapid adsorption of Cr (III) in comparison to the slow reduction and adsorption of Cr (VI) allowing greater soil solution concentrations of the latter species (see section 3.1.6.1.). In comparison, sandy soil, by virtue of its low ion exchange capacity and organic matter content, exerts little effect on added solutions of Cr (III) and Cr (VI) and both species will remain in solution, the former due to lack of adsorption and the latter due to lack of adsorption and reduction. Hence, Soane and Saunder (1959) identified 5 ug/g of Cr (VI) as the toxic threshold for tobacco (***Nicotiana tabacum***) on this soil type compared to a similar amendment of 8 ug/g of Cr (III) recorded as a toxic limit for sugar beet (***Beta vulgaris***) (Hewitt, 1953).

The importance of soil pH was demonstrated by Mortvedt and Giordano (*1975*) who found that maize (***Zea mays L.***) grown in soils of pH 5.5 and 7.0 containing 80 ug/g of added Cr (VI) suffered forage yield reductions of approximately 27 and 52 % respectively. Yield was not significantly affected by an identical application of Cr (III) to soil of pH 5.5.

These findings are clearly in line with the known mobilities of these chromium species in acid and alkaline soils (section 3.1.6.1.). In soil containing 320 ug/g of added Cr (VI), yield reductions were virtually 100 % at either pH, whereas an identical Cr (III) soil concentration resulted in a relatively small yield reduction of 54 % at pH 5.5 (Mortvedt and Giordano, 1975). In a similar study, M^{c}Grath (1982) examined the growth of oat plants (***Avena sativa***) over 35 days in acid, pH 3.9, and alkaline soils, pH 7.6, amended with 750 ug/g of Cr (III) or Cr (VI). In both soils with added Cr (VI) all plants died whereas in the soils with added Cr (III) the plants survived but were stunted and the development of lateral roots was inhibited. In soils amended with Cr (VI) all the chromium in the alkaline soil solution was in the form of Cr (VI), whereas in the acid soil solution, the total chromium concentration amounted to only 15.5 % of that in the alkaline soil and a much smaller fraction of the total was in the form of Cr (VI). Of the plants supplied with Cr (III), symptoms were accentuated in the acid soil, where the total chromium concentration in the soil solution was 40 % greater than in the alkaline soil. Cr (III) additions resulted in much lower concentrations of total chromium in the soil solutions; approximately 7 and 1 %, of the soil solution concentrations from Cr (VI) amended acid and alkaline soils respectively. Again, these findings are in accordance with the behaviour of both forms of chromium in soil (section 3.1.6.1.).

Severe phytotoxic symptoms are frequently found in plants which do not contain appreciably more foliar chromium than plants grown in soils containing normal background levels of chromium (Mortvedt and Giordano, 1975; Cary *et al*, 1977 a), as the roots are the main site of chromium accumulation (Lyon *et al*, 1969; Skeffington *et al*, 1976; Cary *et al*, 1977 a; Lahouti and Peterson, 1979; Anon, 1981; Cary, 1982) irrespective of the form of chromium supplied (Cary *et al*, 1977 a; Skeffington *et al*, 1976). Lahouti and Peterson (1979) for instance, noted that approximately 98 % of chromium uptake was retained in the roots of 9 crop species, while Skeffington *et al* (1976) observed a 100 fold drop in concentration across the hypocotyl of barley seedlings (***H. vulgare***). The main phytotoxic action of chromium therefore appears to occur, via impaired root function, through

interference with plant uptake of essential elements.

For instance, Hewitt (1953) noted symptoms of iron chlorosis in ***B. vulgaris*** due to Cr (III) concentrations in sand of as little as 8 ug/g. These symptoms were corrected by painting the leaves with a solution of $FeSO_4$. Turner and Rust (1971) observed that Cr (VI) concentrations of more than 0.1 ug/cm^3 in nutrient solution was sufficient to reduce levels of iron, calcium, potassium, phosphorus and manganese in the leaf tips, and levels of iron, magnesium, phosphorus and manganese in the roots of soy bean (***Glycine max***). Hunter and Vergnano (1953) found that Cr (VI) at a concentration of 2 ug/g in sand culture increased nickel uptake and specific symptoms of nickel toxicity in ***A. sativa***. These symptoms closely resemble the pale yellow leaf stripes of iron deficiency in cereals and Mengel and Kirkby (1982) surmised that iron chlorosis may be caused by increased nickel uptake, as readily formed nickel chelates can replace other heavy metals from physiologically important sites.

The aforementioned findings indicate that phytotoxic effects, due to the levels of soil chromium found around Rentex treated timber (section 2.3.6., table 2.3.6.5, part B and section 2.3.8., table 2.3.8.1, part B), may occur. Given the greater toxicity of Cr (VI) in soil, due to its enhanced availability in the soil solution (Mortvedt and Giordano, 1975; M^{C}Grath, 1982), and its transient nature in soil and treated wood (section 3.1.6.1.), the most severe of any phytotoxic effects, due to the elevated soil concentrations of total chromium around treated timber, are likely to occur within a short timescale after remedial treatment.

3.1.6.2.2. Bioaccumulation and bioavailability of chromium.

Foliar accumulation of chromium from concentrations in uncontaminated soil is poor due to the normally low levels of chromium in the soil solution, by virtue of the generally low solubility and mobility of the dominant Cr (III) species (see section 3.1.6.1.), and the containment of the bulk of absorbed chromium in the root (section 3.1.6.2.1.). As

concentrations of chromium in United Kingdom soils typically range from a mean of 34 ug/g in England and Wales to a mean of 62 ug/g in Scotland (M^{c}Grath, unpublished, cited by M^{c}Grath and Smith, 1990; Berrow and Reaves, 1986) low tissue chromium concentrations of up to only a few ug/g are generally found over a range of plants (Anon, 1981; Mengel and Kirkby, 1982; M^{c}Grath and Smith, 1990). Even in serpentine soils which can contain levels of total chromium as high as 125,000 ug/g (Shewry and Peterson, 1976), concentrations of chromium in non-accumulator plants are rarely in excess of 100 ug/g (Shewry and Peterson, 1976; Jaffre *et al*, 1979; Brooks and Yang, 1984), and concentrations of chromium in plants grown on soils containing chromium loaded wastes such as fly ash (Adriano *et al*, 1980) and sewage sludge are barely above background levels (Mortvedt and Giordano, 1975; Cary, 1982) reflecting the formation of very stable organic complexes or precipitates with Cr (III) (M^{c}Grath and Smith, 1990).

Given these low plant uptake rates of soil chromium and the status of chromium as an essential trace element for normal mammalian carbohydrate metabolism (Anderson, 1981; Starich and Blincoe, 1983), studies have been carried out to examine chromium concentrations within various crop plants and methods for increasing these concentrations in crop plants for human and animal consumption (Cary *et al*, 1977 a, b; Lahouti and Peterson, 1979; Ramachandran *et al*, 1980). In general terms, the foliage of leafy vegetables such as spinach (***Spinacea oleracea***) and cauliflower (***Brassica oleracea***) tended to accumulate more chromium than cereals, cereal grains being particularly resistant to chromium accumulation (Cary *et al*, 1977 a; Lahouti and Peterson, 1979; Ramachandran *et al*, 1980). These studies and others (Skeffington *et al*, 1976; M^{c}Grath, 1982) also identified higher chromium concentrations in the shoots of plants supplied with Cr (VI) in solution culture. This appears to be due to the binding of Cr (III) by cation exchange sites in the cell walls (Skeffington *et al*, 1976; Lahouti and Peterson, 1979) which restricts the passage of this species through the root cortex and its transport in the xylem (Skeffington *et al*, 1976).

However, Cary *et al* (1977 b) concluded that due to the reduction of soluble Cr (VI) to insoluble Cr (III) in soils (see section 3.1.6.1.) the addition of Cr (VI) to soil would be very inefficient as a method for increasing the chromium content of food crops. Other studies have indicated that substantial increases in foliar chromium concentrations can be induced by additions of Cr (VI) or Cr (III) to soil (Shewry and Peterson, 1974; Mortvedt and Giordano, 1975; M^{c}Grath, 1982). However these increased foliar concentrations were only evident at soil chromium concentrations which were demonstrably highly phytotoxic.

For instance, Mortvedt and Giordano (1975) found foliar chromium concentrations of 29 and 95.3 ug/g in maize (***Z. mays***) due to Cr (VI) applications of 320 ug/g to acid and alkaline soils respectively. These values compared with 2.5 and 3.1 ug/g in plants due to a Cr (VI) addition of 80 ug/g to acid and alkaline soils respectively and 1.6 ug/g in plants from a control soil of acid pH but were accompanied by an almost 100 % forage yield loss. A chromium addition of 320 ug/g as Cr (III) to the acid soil resulted in a 54 % forage yield reduction due to a foliar chromium concentration of only 2.8 ug/g compared to the control value (Mortvedt and Giordano, 1975), whereas oat plants (***Avena sativa***) grown for 35 days in acid and alkaline soils amended with 750 ug/g of Cr (III) possessed foliar chromium concentrations of 144 and 15 ug/g respectively compared to 0 ug/g in respective control plants but root and shoot dry weights were reduced to between 13 and 20 % of control plant values (M^{c}Grath, 1982).

Therefore, even accepting that excessive concentrations of chromium (VI) are introduced into the soil and soil solution adjacent to remedially treated distribution poles, lethal toxic effects on the surrounding vegetation (section 3.1.6.2.1.) would most likely preclude foliar accumulation. Alternatively, foliar accumulation may not occur because soil conditions are unlikely to favour the presence of Cr (III) in the soil solution (section 3.1.6.1.). The concentration of both chromium species in the soil solution is also likely to be diluted by leaching. The bioavailability of elevated chromium concentrations in soils adjacent to Rentex treated timber (sections 2.3.6. and 2.3.8.) via accumulation in vegetation

is therefore likely to be very poor. In addition, direct animal consumption of, or contact with, chromium in soil is unlikely to provide an entry for chromium into terrestrial ecosystems as inorganic forms of chromium are poorly absorbed in the gut (Starich and Blincoe, 1983). However these chromium concentrations are likely to pose a greater threat to soil micro-organisms, and the processes they mediate, by virtue of the greater direct exposure to which these organisms are subjected.

3.1.6.2.3. Some toxic effects of chromium on soil micro-organisms.

Though chromium soil additions of 1 ug/g can stimulate soil bacterial numbers (Zibilske and Wagner, 1982), chromium additions to soil invariably have a negative effect on soil micro-organisms. Drucker *et al* (1979), cited by Wong and Trevors (1988), found that Cr (III) added to soil at concentrations of as little as 10 ug/g significantly reduced the numbers of aerobic and anaerobic bacteria, and a Cr (VI) concentration of 10 - 12 ug/cm^3 was sufficient to inhibit most soil bacteria isolates in broth culture (Ross *et al*, 1981). The greater toxic effect of Cr (VI) was demonstrated by Ross *et al* (1981) who found that respiration in soils was decreased by chromium additions of 10 or 100 ug/g of Cr (VI) or Cr (III) respectively. The findings of Drucker *et al* (1979) indicate that fungi are less sensitive to the effects of chomium in soil, with depression of fungal populations occurring at a Cr (III) concentration ten times that required to reduce bacterial numbers. Cr (VI) was found to be more toxic to fungi than Cr (III) with mycelial growth rates more inhibited by Cr (VI) than Cr (III) (Babich *et al*, 1982).

Given these indications of the sensitivity of soil bacteria to chromium it is not surprising that Ajmal *et al* (1984) found that electroplating wastes rich in Cr (VI) were toxic to nitrifying bacteria. The nitrifying bacteria carry out the oxidation of ammonia -> nitrite -> nitrate, the first step mediated by species including ***Nitrosomonas***, ***Nitrosolobus*** and ***Nitrospira***, and the second by ***Nitrobacter*** species. These bacteria are therefore of great importance with respect to plant available nitrogen in soil. However, a severe reduction in

the rate of nitrification is unlikely to be a feature of the soils subjected to the degree of chromium contamination arising from Rentex remedial treatment (sections 2.3.6. and 2.3.8.) as James and Bartlett (1984) found that nitrate production in soils amended with 520 or 5200 ug/g of chromium as Cr (VI) or Cr (III) was not significantly different from that in uncontaminated soil. Nitrate formation was only significantly reduced during the first 23 days after a soil amendment of 52,000 ug/g, and 21 days later nitrate levels had returned to control soil values.

These results indicate that the chromium levels found in field soils adjacent to preservative treated distribution poles may inhibit the indigenous microbial population.

3.1.6.3. Some toxic effects of chromium in aquatic environments.

Possible harmful effects of leached preservative chromium within terrestrial ecosystems (section 3.1.6.2.), are mediated by those soil conditions which enhance the presence of either chromium species in the soil solution (sections 3.1.6.1., 3.1.6.2.1. and 3.1.6.2.2.). These soil conditions will also favour the leaching of chromium from the soil into the groundwater and its entry into aquatic environments. Given the restricted mobility of Cr (III) in soil (section 3.1.6.1.), the movement of this species to and through the groundwater will probably not occur to any great extent. However, the mobile Cr (VI) species (section 3.1.6.1.) is likely to be leached into groundwater, and provided the groundwater is shallow and situated below a free draining soil profile, allowing oxygen replenishment to maintain oxidising conditions and slow its reduction, this species is capable of extensive movement. Calder (1988) for instance, described groundwater plumes of Cr (VI) extending up to 1,500 metres from permeable surface sites used for the disposal of electroplating, mining and CCA (copper chrome arsenate) preservative wastes.

Despite these indications of chromium mobility and the evident biomagnification of heavy metals including chromium in certain marine dwelling organisms (Lande, 1977; Guthrie *et al*, 1979), as indicated for fluoride (section 3.1.5.2.) it is extremely unlikely that chromium, introduced into the soil via the remedial treatment under study, could seriously threaten the marine environment. However, freshwater systems will be more accessible and at greater risk.

Growth inhibition of the freshwater alga ***Ulothrix fimbriata*** was recorded at a Cr (VI) concentration of only 0.15 ug/cm^3 whereas ***Cladophora glomerata*** and ***Stigeoclonium tenure*** required a Cr (VI) concentration of 0.25 ug/cm^3 for inhibition to occur (Bharti *et al*, 1979). Petria (1978) found a ***Chlorella*** species to be more resistant with 10 ug/cm^3 as Cr (VI) the toxic threshold for inhibition of growth in this alga, though a reduction in photosynthesis was noted within a Cr (VI) concentration range of 0.05 - 0.100 ug/cm^3. These variations in sensitivity can have important effects on species composition within freshwater algal communities. For instance, a chromium addition of 0.4 ug/cm^3 as Cr (VI) caused a shift in community dominance from diatoms to blue-green and green algae (Patrick, 1978).

The freshwater crustacean ***Daphnia magna*** is as sensitive to chromium concentrations as algae with Biesinger and Christensen (1972) reporting a 16 % reproductive impairment due to 0.33 ug/cm^3 as Cr (III), while toxic thresholds due to Cr (VI) range from 0.016 - 0.70 ug/cm^3 (McKee and Wolf, 1963). However, insects seem to have greater tolerance with 96 hour LC50s in the range of 43 - 64 ug/cm^3 as Cr (III) for a number of fly larvae and nymphs (Anon, 1976).

These organisms make up the prey species for many freshwater fish, which themselves can be very sensitive to chromium. Olson (1958) found that salmon fingerlings exposed for 12 weeks to a chromium concentration of 0.20 ug/cm^3 as Cr (III) were not affected, however a similar period of exposure to Cr (VI) at the same concentration resulted in 53 %

mortality. A sharp retardation in the growth of fish exposed for 2 weeks to 0.20 ug/cm^3 as Cr (VI) was recorded and this effect was accentuated by reducing the water temperature.

3.1.7. The environmental impact assessment of Rentex remedially treated timber.

3.1.7.1. Natural indicators of harmful environmental effects and the challenge in measuring these effects.

Some of the possible environmental hazards associated with significant increases in soil concentrations of fluoride and chromium, like those in the soils adjacent to Rentex treated timber at the Glenclova, Tealing and Oban field sites (section 2.3.6., table 2.3.6.5, parts A and B, and section 2.3.8., table 2.3.7.1, parts A and B), noted in the literature review (sections 3.1.5. and 3.1.6.), clearly indicated the requirement for investigations to determine the environmental impact of remedially treated timber in use. The literature survey (sections 3.1.5. and 3.1.6.) also identified those natural indicator systems (section 3.1.4.), within the domain of treated timber, at which these investigations should be aimed. These are: plants, to identify any toxic symptoms and/or bioaccumulation in the shoots; drainage waters, to determine the specific toxic chemical species leached from treated wood and their possible movement from the area of soil contamination into groundwater; and microbial activity, to evaluate potentially damaging effects on important soil processes.

To carry out such studies in the field presented a variety of problems. Firstly, all of the field sites used in the experiments to determine preservative efficacy (sections 2.2.2. - 2.2.7.) were subject to grazing and poaching by farm stock, and at sites where these animals were excluded, the extreme grazing activities of the local rabbit populations were all too evident. Plant material isolated from environmental effects other than the presence of treated timber, was therefore largely unavailable. Secondly, the collection of groundwater samples via the excavation of boreholes around poles would present problems due to the

difficulties in collecting such samples without contamination via the migration of soil contaminants down the well bore or by the entry into the borehole of contaminated soil from above (Graham, 1991). In any case a simple set of groundwater samples would not provide a detailed characterisation of contaminant concentration, speciation and mobility in the soil leachates nearer the soil surface, ie. those concentrations available to vegetation. Lastly, the removal of field soil samples for activity measurements of soil micro-organisms would be problematical considering the difficulties already experienced in removing very much smaller soil samples for chemical analysis (section 2.2.5.). These soil operations would also have interfered with the aforementioned sampling operations.

In order that accurate sampling of plant material would not be hindered by grazing causing periodic removal or absence of plant cover, a new protected field site was required. To sample groundwater and shallower soil waters close to treated timber, without contamination problems, the design and installation of an extensive drainage system would be necessary. In addition, to facilitate these installations and for adequate soil volumes to be removed for microbial activity measurements, the chosen site would require to be easily worked and free of stones. In view of the cost, time and considerable site disruption which would be incurred by these operations, the assessment of specific environmental effects of treated timber in the field was not regarded as a feasible proposition. However, in view of the necessarily artificial nature of small scale bench top laboratory experiments which might have been attempted to satisfy these assessments, this approach was considered as being too unrepresentative of field conditions.

3.1.7.2. The use of physical field models in eco-toxicological studies and the development of a model for environmental impact assessment of Rentex remedially treated timber.

The constraints associated with the environmental assessment of Rentex remedial treatment (section 3.1.7.1.) are common to a variety of eco-toxicological studies and many workers have constructed physical model systems to overcome them. As defined by Jorgensen (1990), physical models contain the main components of the real system in an attempt to observe processes and reactions of the complex field system within the confines of the simpler model system. Models retain the key benefits associated with both field and laboratory experiments in that the realistic nature of the former is linked to the degree of control and accessability associated with the latter.

Aquatic and semi-aquatic laboratory based model ecosystems have been used by a number of workers to evaluate environmental effects of pesticides in freshwater (Reinert, 1972; Metcalf, 1974; Isensee, 1975). Reinert (1972) studied the accumulation and biomagnification of dieldrin in algae (***Scenedesmus obliquus***), the water flea (***Daphnia magna***) and the guppy (***Poecilia reticulata***), when various concentrations of the pesticide were added to static waters in aquaria of 4.5 litre. The amount of pesticide accumulated by each species was directly proportional to its concentration in water and when ***P. reticulata*** were fed daily rations of ***D. magna*** containing dieldrin at different concentrations the amounts of dieldrin accumulated by the fish were directly proportional to those concentrations in ***D. Magna***. Isensee (1974) improved on this system by constructing flowing and static aquatic ecosystems of up to 80 litre capacity with more trophic levels. In these models pesticides were first mixed with soil to allow adsorption before addition to the water to simulate pesticide entry into aquatic ecosystems via erosion of pesticide treated soil from agricultural land. The laboratory model ecosystem constructed by Metcalf (1974) represented a further advance towards real field conditions. This model consisted of an aquarium containing a shelf of washed white quartz sand sloping down to a 7 litre 'lake' of

mineral rich water, which provided nutrition for plants on the shelf and for algae and other aquatic life in the 'lake'. In order to simulate the entry into 'soil' and water of pesticides applied by normal crop spraying practices, the plants seeded on the shelf were treated with a radiolabelled test compound at a standard pesticide application rate.

Though these examples (Reinert, 1972; Metcalf, 1974; Isensee, 1975) demonstrate developments aimed at the construction of models more representative of field conditions, these models must be regarded as basic with respect to environmental impact assessment, lacking scale, effective pesticide dissipation effects and target organism avoidance mechanisms. However, a similar system has been usefully employed in toxicological studies of wood preservatives. Wegen (1990) examined the leaching of a chromium/copper salt wood preservative in water contact. Leaching of preservative constituents decreased as the time interval between wood treatment and water contact increased, due to increasingly effective preservative fixation in the wood (see section 2.1.5.). As might be expected, toxic symptoms in fish, maintained in tanks receiving the leach waters, were proportional to the amount of preservative leached.

With regard to a physical model designed to accommodate the specific environmental systems identified as probable indicators of harmful effects associated with remedially treated timber (section 3.1.7.1.), a terrestrial based model would be more suitable and these have been studied in more detail.

Beall *et al* (1976) described the design of a glasshouse agro-ecosystem, initially used to monitor pesticides in the atmosphere around sprayed crops. Crops were grown on 15 cm of soil within glass cases with an internal volume of 0.75 m^3. Sprinklers within the cases were used to spray pesticides at standard application rates, and the pesticides remaining in the atmosphere around the crop canopy were extracted through vents containing polyurethane filters, which trapped the pesticides for analysis. Though this agro-ecosystem was shown to be an effective research tool, its focus on atmospheric pesticide concentrations obviously

limits its relevance to the design of a model mainly concerned with soil contamination, as in the case of Rentex remedial treatment.

Perhaps the best known and researched physical model systems which can be used for assessments of soil contamination are lysimeters. Lysimeters consist of large encased soil cores on which crops can be cultivated almost as in the field and from the base of which water draining through the profile can be collected and analysed for any leached pollutant. At Rothamstead in the United Kingdom for instance, lysimeters are used to measure the leaching of nitrate, phosphate, potassium and pesticides through agricultural soils (Goulding and Poulting, 1992).

Early lysimeter constructions, circa 1971, consisted of shallow stainless steel containers which were infilled with field soils and maintained above ground (Steffens *et al*, 1992). However, the convenience of the infilling technique was discarded as it became clear that the physical and chemical properties of such homogeneous soils were quite different to heterogeneous field soils, particularly with regard to the partitioning of solutes between mobile and stationary phases, and water movement (Belford, 1979; Hance and Fuhr, 1992), as the latter soils possess preferential flow paths, due to structural cracks and wormholes (Steffens *et al*, 1992), and spatial variability in hydraulic properties (Hance and Fuhr, 1992). Hence, the homogeneous nature of disturbed soils may give rise to findings which bear little relationship to what occurs in the field. A good example of this is the oxidation of chromium (III) to chromium (VI) found in homogeneous soil samples (James and Bartlett, 1979) but not in heterogeneous field soils (Bartlett and James, 1988) (see section 3.1.6.1.).

Therefore, in order to preserve the physical properties of heterogeneous field soils, lysimeters now consist exclusively of enclosed soil monoliths removed intact and undisturbed from the field, with surface areas of up to 1 m^2 and depths of up to 2 m. They are extremely adaptable tools allowing accurate and reproducible crop, soil and leachate measurements in any soil type, in the glasshouse or in the field, and can be maintained above

or below ground (Belford, 1979; Figge, 1992; Scholz *et al*, 1992; Steffens *et al*, 1992; Yon, 1992). One major disavantage to the use of lysimeters is the extensive excavations required to recover and install them (Belford, 1979; Traub-Eberhard *et al*, 1992), and consequently these model systems are expensive, costing at least £2,000 each (Goulding and Poulting, 1992).

However, similar systems have been used in environmental studies of wood preservatives. Bergholm (1992) used lysimeters buried in the field to examine the mobility of copper, chromium and arsenic leached from wood chips of Boliden-K33 (CCA) treated timber, in order to evaluate soil burial as a safe disposal strategy for such timber wastes. The lysimeters were 30 cm in diameter and 70 cm deep and were infilled with a 35 cm layer of soil covered with a 15 cm deep layer of wood chips and a top layer of 5 cm of the same soil. Three lysimeters were constructed using clay, peat and sandy soils. Leachate was collected at the base of each lysimeter in an experiment which was pursued for 11 years. Of the concentrations of each element leached from the wood chip layer more than 90 % was retained in the soil columns and as might be expected both arsenic and chromium were more mobile in the sandy soil. However, findings from lysimeter studies (Belford, 1979; Hance and Fuhr, 1992) indicate that the homogenised soils used by Bergholm (1992) would not be representative of the heterogeneous field soils in which these timber wastes would be buried.

In contrast, with respect to the remedial treatment in the present study, preservative injection is preceded by a soil excavation around the pole to a depth of approximately 0.5 - 0.75 m at the groundline of each distribution pole (section 2.2.3.2.1.), which procedure effectively destroys the heterogeneity of the soil in this zone. The use of a standard lysimeter, consisting of a heterogeneous undisturbed soil monolith, as a physical field model for environmental impact studies of Rentex treated timber was therefore rejected in favour of a homogenised field soil which would be more representative of the field conditions encountered by leached preservative constituents. The use of a homogeneous field soil in

the model construction (section 3.2.2.1.) facilitated the placement of a series of simulated field drains within the soilbed (section 3.2.2.2.) for leachate collection to assess pollutant mobility around treated timber and potential for movement into groundwater. This method of leachate collection reduced the possibility of artificial leachate contamination due to the physical movement of contaminated soil, which is commonly found in borehole examinations of groundwater (Graham, 1991).

In a smaller scale laboratory based environmental study of copper chromium arsenic preservatives, Murphy and Dickinson (1990) placed treated wooden stakes in small shallow containers of different soil type and pH, which were periodically flooded with waters of different pH. Chemical analysis of leachates and soils indicated accumulation of preservative components, with concentrations varying according to experimental conditions, component type and preservative formulation used. This study therefore provided useful information on the relative leach resistance of several wood preservative formulations. However, as a design for environmental assessments of these preservatives it is of doubtful value as the size and condition of the stakes was unrepresentative of treated field structures, and bioassays which may have confirmed the environmental hazard potential of leached preservative constituents were not undertaken.

The physical field model designed for the present study contained aged and scaled down pole sections which had been remedially treated by the normal injection procedure (section 3.2.2.4.). For the assessment of phytotoxic and bio-accumulatory effects adjacent to treated timbers, perennial ryegrass (***Lolium perenne***) was chosen as the primary crop species for cultivation in the physical model (section 3.2.3.1.). This grass is known to be sensitive to chromium (Breeze, 1973), and it is of great economic importance as the principal grass variety within grasslands, which predominated adjacent to the remedially treated field poles used in the efficacy studies (sections 2.2.2. - 2.2.7.), and which account for up to 75 % of the more intensive agricultural land in the United Kingdom (Holmes, 1980).

The physical field model, the construction and operation of which is detailed later (sections 3.2.1. - 3.2.4.), therefore contained all the natural systems identified as being probable indicators of the environmental impact of remedially treated timber (section 3.1.7.1.), and allowed their study under controlled conditions in the laboratory. An artificial rainfall regime was supplied to the entire surface area of the field model via an overhead sprinkler apparatus (section 3.2.2.5.), and to provide severe leaching conditions based on actual rainfall patterns, a leaching rate acceleration factor was built into this regime by consulting relevant rainfall records (section 3.2.4.2.).

3.1.8. Brief outline of the experimental programme to assess the environmental impact of Rentex remedially treated timber using a physical field model.

The natural systems determined to be at greatest risk from the presence of remedially treated timber in the field, due to the leaching of the toxic preservative constituents fluoride and chromium, namely plants, drainage waters and microbial activity (section 3.1.7.1.), will be used as the main indicators of any harmful environmental effects associated with remedially treated timber in the field. In view of the constraints which prevent the accurate monitoring of these indicator systems in the field (section 3.1.7.1.), a laboratory based physical field model was designed to study these systems under controlled conditions (section 3.1.7.2.). Three models were constructed, including one control (sections 3.2.1. - 3.2.3.), and the following studies were undertaken:

(1) Samples of simulated rainfall applied to the model units (section 3.2.4.2.) and leachate drained from the soilbed of each model unit were collected (section 3.2.4.3.) and analysed for fluoride, total chromium and chromium (VI) content (section 3.2.5.1.). The objective of this experiment was to identify the presence of these chemicals in soil waters adjacent to Rentex treated timber; to determine the extent to which these chemical concentrations were altered by changes in soil

texture; and to evaluate the groundwater contamination risk associated with the remedial treatment.

(2) Crops of perennial ryegrass (***Lolium perenne***) and rye (***Secale cereale***) were grown in each soilbed (sections 3.2.4.4. and 3.2.4.5. respectively) and were subjected to density and yield measurements (sections 3.2.4.4. and 3.2.5.2.1., and section 3.2.4.5.5. respectively), to evaluate any effects on these plant parameters due to the proximity of remedially treated timber. In addition, sward samples of ryegrass were analysed for fluoride and chromium content (section 3.2.5.2.2.), to identify any accumulation of these preservative constituents in vegetation adjacent to treated timber.

(3) Soil samples were recovered from each soilbed (section 3.2.4.6.) and analysed for fluoride and chromium content (section 3.2.5.4.1.) to identify any soil contamination around remedially treated timber due to the leaching of these preservative constituents. Sub-samples of a proportion of those soil samples used for chemical analysis (section 3.2.4.6.) were monitored for microbial activity (section 3.2.5.4.2.) to evaluate any potential deleterious effects of remedially treated timber on important soil processes, due to elevated soil concentrations of fluoride and chromium.

(4) Remedially treated pole sections erected in the soilbeds were recovered and wood samples removed from the preservative treated groundline area (section 3.2.4.7.) for chemical analysis of fluoride and chromium content (section 3.2.5.5.) to identify any reduction in the concentrations of these preservative constituents within the treated timber.

The success of the physical field model in providing an accurate assessment of the environmental mobility and eco-toxicity of the preservative elements leached from

remedially treated 'in-service' distribution poles will be determined by a series of critical comparisons between parameters common to both the developed model and the field studies presented in chapter 2. These comparisons will establish the suitability of the designed model for inclusion in wood preservative testing protocols.

3.2 MATERIALS AND METHODS

3.2.1. Brief outline of model system.

Each model unit consisted of a 2 m long distribution pole section positioned vertically in a grass covered sandy loam soil bed which contained a number of simulated field drains for leachate collection. An overhead tapwater misting apparatus provided simulated rainfall and lighting was provided on a day/night cycle. Three model units were prepared, two containing Rentex treated creosoted pole sections and one containing a creosoted pole section untreated with Rentex. A lighter textured soil was used for one of the soilbeds containing a treated pole section.

3.2.2. Model preparation.

3.2.2.1. Soil beds.

A free draining sandy loam soil was obtained from the Scottish Crop Research Institute at Dundee. The soil was collected from the upper 15 cm of topsoil from a field site which had received no previous chemical treatment and was stored outside in covered plastic bins until required.

A stony base for each soil bed was produced by utilising that fraction of the soil failing to pass a 1 cm mesh stainless steel sieve. Soil fractions which passed and failed to pass a 0.5 cm mesh stainless steel sieve were used as topsoil and subsoil respectively. To enhance soil drainage characteristics, topsoil and subsoil fractions were amended by the addition and thorough mixing of 1 part washed aquarium gravel to 3 parts soil by volume.

Soil profiles were constructed within three 227 dm^3, high density polyethylene water tanks of dimensions 55 cm x 55 cm x 108 cm. The layers of each profile were given a 2^0 slope (figure 3.2.1) to represent prevailing ground conditions at the largest Rentex field site at Glenclova (sections 2.2.3., 2.2.4. and 2.2.5.). The topsoil and subsoil of 1 model unit was further amended by the addition and thorough mixing of 1 part washed sand to 2 parts soil by volume.

3.2.2.2. Drainage system.

During the construction of each soil profile, simulated field drains were placed at various levels (figure 3.2.2). The horizontal distance between each drain, excluding drain 4, was 20 cm. The vertical distance between drains, excluding 4 and 9, was 12.5 cm. Drain 4 was positioned in contact with the interior surface of of the tank, just below the soil surface.

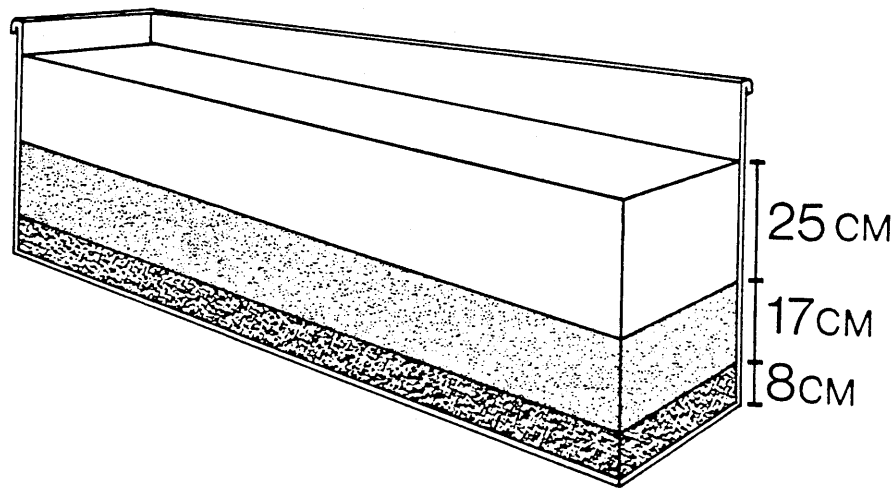


Figure 3.2.1. Exposed section of soil tanks showing sloping soil profiles constructed from sieved fractions of the original field topsoil.

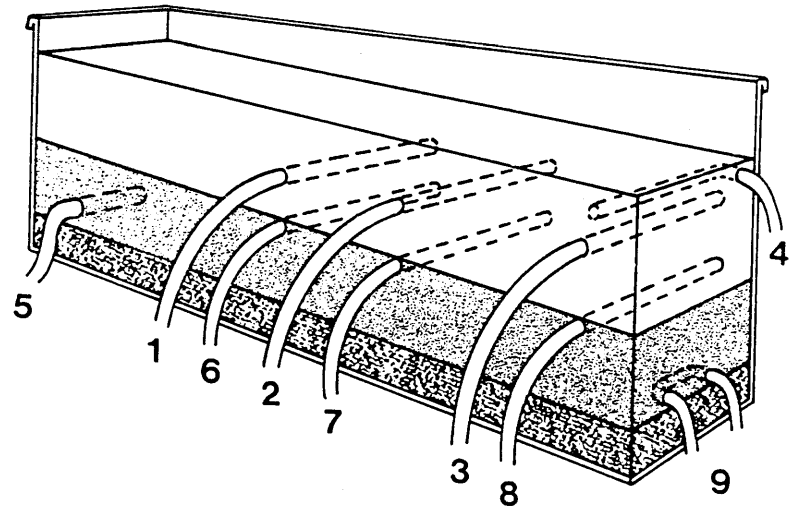


Figure 3.2.2. Arrangement of numbered drains postioned within each soil profile for leachate collection.

Each drain consisted of flexible 12 mm bore polyvinyl chloride (PVC) piping, pierced on the upper surface with 9 x 4 mm diameter holes for every 50 mm of length (figure 3.2.3). To facilitate the entry of leachate waters and prevent blockage by soil, each drain was topped with a permeable 30 mm deep layer of washed aquarium gravel. Figure 3.2.4 shows a side elevation of the soil bed detailing drains 1 - 3 and 6 - 8. The continuous gravel layer above the lower drains 6 - 8 was designed to channel a broad front of drainage water from above to these drains to prevent flooding of the soil bed.

Each drainage port was fitted into a covered plastic container. The volume of each container was dictated by the likely volume of leachate each drain would receive and ranged from 4.5 dm^3 for drains 1,2,3 and 4, to approximately 50 dm^3 for drain 9. For the soil bed containing the untreated pole section (section 3.2.2.4.), the ports of drains 5 and 9, and 1, 2, 3, 4, 6, 7 and 8 were combined for flow into containers of approximately 50 and 20 dm^3 capacity respectively. A 50 dm^3 capacity container was provided for collection of a portion of simulated rainfall (section 3.2.2.5.) falling outside the soil beds, via corrugated plastic gutters extending between the soil beds.

At no time during the experimental period (section 3.2.3) was leachate produced from topsoil drains, 1, 2, 3 and 4, of any model unit, as improvements to the soils' drainage characteristics (section 3.2.2.1.) had rendered the topsoil less resistant to water flow than the drains it contained.

3.2.2.3. Chemical characterisation of soil.

Topsoil and subsoil removed from the top of each soil bed slope, to allow pole section insertion (section 3.2.2.4.), was collected and homogenised for each model unit. Representative samples of each, obtained by spreading and quartering (section 2.2.6.4.), were measured for fluoride and chromium content as for field soils (section 2.3.4.). The

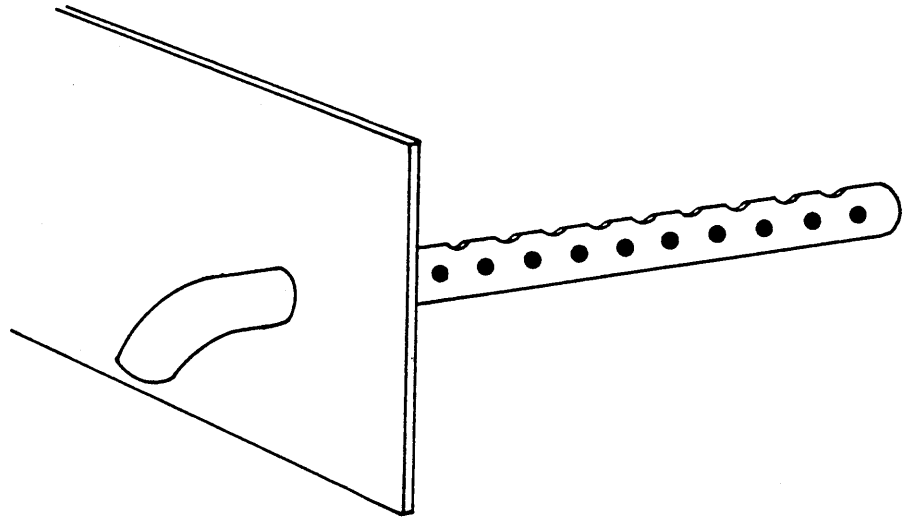


Figure 3.2.3. Detail of PVC piping drain with holes to receive water flow from surrounding soil.

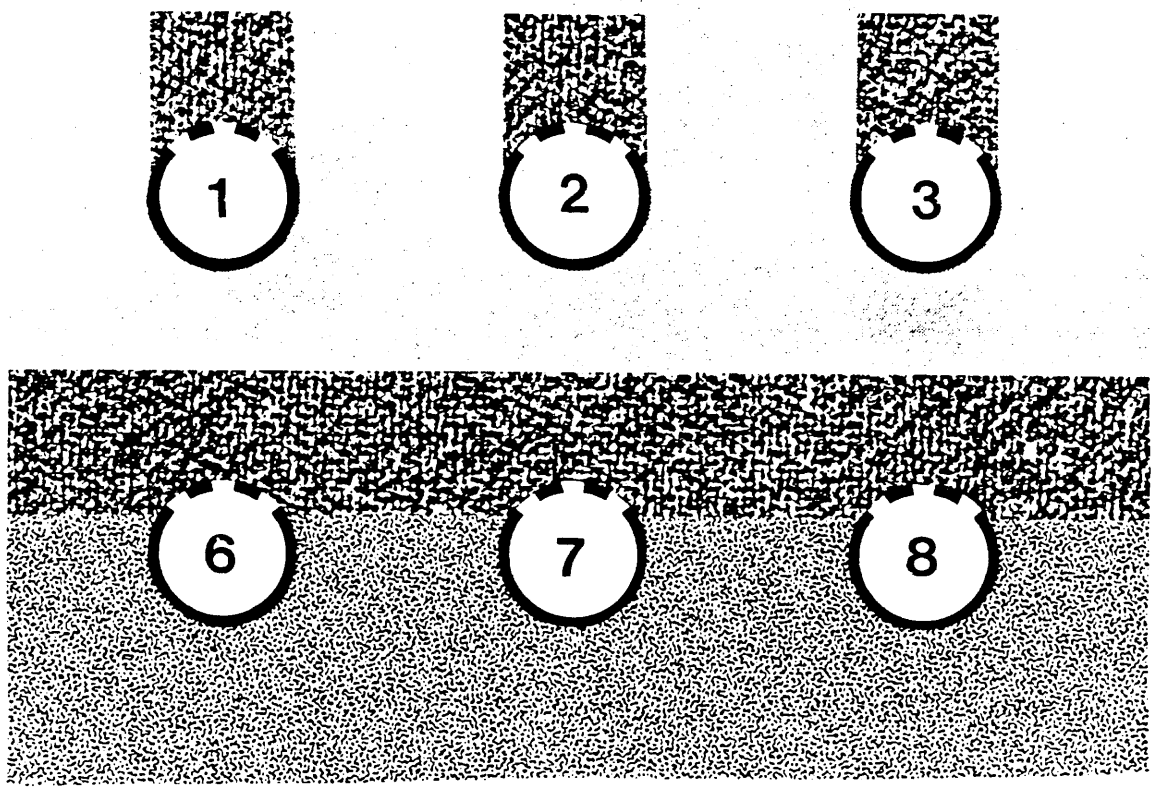


Figure 3.2.4. Side elevation of soil profile drains detailing drainage holes and permeable gravel layers.

soils which had received no addition of sand were combined and homogenised again. This provided 2 homogenised samples, 1 soil and 1 soil/sand mixture. Representative samples of each, obtained by spreading and quartering (section 2.2.6.4.), were used for measurement of cation exchange capacity and organic matter content by standard methods (Anon, 1986 c), water holding capacity according to the method of Avery and Bascombe (1974) and pH (BS 1377: 1975) (see table 3.2.1).

3.2.2.4. Pole sections.

Six 2 m long pole sections cut from the upper portions of aged creosoted distribution poles of Scots pine were prepared for insertion into the soil beds. The percentage moisture contents of 4 pole sections, of approximate diameter 15.5 cm, were measured at the groundline position as for field poles (section 2.2.3.), then each was Rentex treated by the injection process (section 2.1.2.), each receiving 63 preservative injections. Bitumen was applied to the surface of the treated area which was covered above the groundline with an aluminium sheath. All pole sections were stored indoors for 2 years.

To provide pole section moisture contents similar to those found in field poles the sections were subsequently conditioned for 3 months at a mean relative humidity and temperature of 94% and 24°C respectively, measured using a *Vaisala* HM 34 Temperature and Humidity Meter. The percentage moisture content of each pole section was measured at the groundline position as before and was found, for all pole sections, to lie between 15 % and 19 %, i.e at the lower end of the range of moisture contents found for field poles (section 2.3.4., tables 2.3.4.1 and 2.3.4.2).

Two treated and 1 untreated pole section were dug into the soil bed slopes, the surface of each pole section being approximately 25 cm downslope of the interior surface of the water tank and the base of each pole section resting approximately 5 cm above drain 5

Table 3.2.1. Chemical and physical characteristics of soils from the model units

Analytical Measurement	Soil Type (Model Unit)	
	Sandy Loam Soil	Sand Amended/ /Sandy Loam Soil
pH	6.15	5.45
Cation Exchange Capacity (me/100g)	11.61	5.96
Organic Matter Content (g/kg)	33.6	18.8
Water Holding Capacity (%)	18.21	13.75

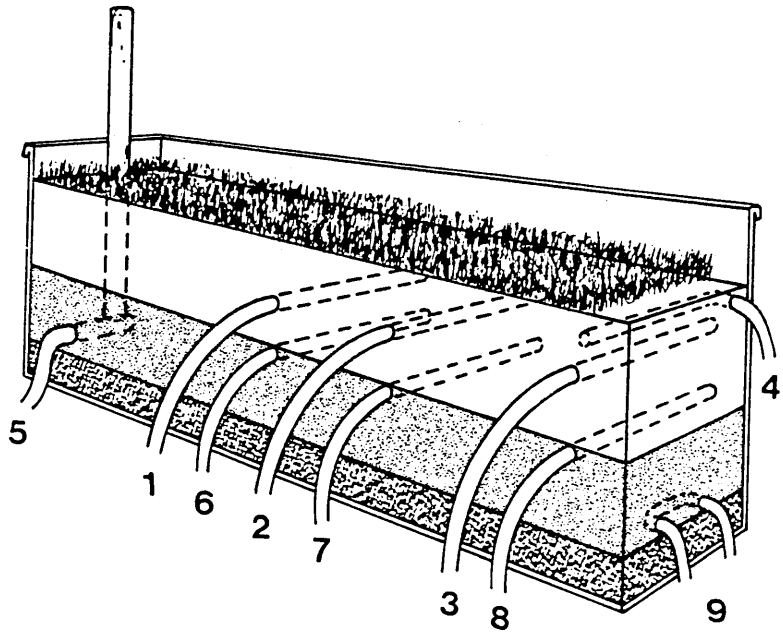


Figure 3.2.5. A treated pole section in place at the top of the sloping soilbed.

(figure 3.2.5). The soilbed amended with sand received 1 of the treated sections. The two further treated sections were set aside nearby and covered, to be subjected to the same conditions of temperature and relative humidity as those positioned in the soil beds (section 3.2.3.3.).

To increase the moisture contents of soil bed pole sections to levels more representative of field poles (section 2.3.4., tables 2.3.4.1 and 2.3.4.2), each was subjected to 2 weeks in contact with a raised watertable approximately 20 cm deep. This was achieved by inverting the ports of drains 5 and 9 (figure 3.2.5) and watering the soil bed and pole section with an overhead tapwater misting unit (section 3.2.2.5.1.) until the outlets of these drains were full.

3.2.2.5. Simulated rainfall.

3.2.2.5.1. Apparatus.

Simulated rainfall was provided for each model soil bed by an overhead tapwater misting unit (Philip Harris Education). Three spray heads were connected in series with polythene pipework, each spray head consisting of an atomiser jet supported 165 cm and 15 cm above each soil bed and pole section respectively by a rigid PVC rise pipe. As the surface area of each soil bed was 0.594 m^2 and the coverage of each atomiser jet was 1 m^2, 2 sheets of polystyrene approximately 0.75 m^2, were slotted between each spray head to prevent cross spraying between soil beds.

3.2.2.5.2. Calibration of spray heads.

Each connected spray head was placed in a separate plastic container of identical volume, approximately 4.25 dm^3. The connected heads were attached to the mains water supply, which was turned fully on. The spray heads operated till each container overflowed,

which occurred simultaneously during a series of 7 such tests, indicating that the flow rate through each head was effectively equal at this water pressure. Therefore the flow rate for each spray head could be determined by measuring the total volume of tapwater produced by all 3 spray heads over a given period of time.

Accordingly, the pipework connecting each raised spray head was supported in a plastic trough 3 m long and each erect spray head was covered with a plastic bag which opened into the trough. The water supply was turned fully on and the combined flow through the heads entered the trough which was tilted to empty into a plastic container. After precisely 5 minutes, the water supply was shut off and the volume of water collected was measured using 100 cm^3 and 1 dm^3 graduated measuring cylinders. This test was replicated 5 times and the mean volume of water produced by the combined spray heads over 5 minutes was 5.518 dm^3 (standard deviation, 0.071), equivalent to 1.839 dm^3 for each spray head. Therefore, the flow rate for each spray head was 0.368 dm^3min^{-1}.

3.2.3. Model conditions.

3.2.3.1. Plants.

Seed of a perennial ryegrass (***Lolium perenne***) variety 'Fennema' and an un-named variety of rye (***Secale cereale***) was recommended by and obtained from Twyford Seeds Limited for inclusion in each model unit (sections 3.2.4.4. and 3.2.4.5.). The ryegrass variety (2 swards of which were grown and sampled before rye was sown) was recommended for its characteristics of disease resistance, good ground cover and persistance under wet upland conditions. Rye was favoured, in preference to barley, for its faster growth and deeper rooting habit, though the complication of a reclining growth habit was a possibility (section 3.2.4.5.2.).

All growing plant material was fed once a week with a standard NPK 10:10:27 liquid fertiliser using a hand held 1 dm^3 capacity mist sprayer.

3.2.3.2. Lighting.

Photosynthetically active radiation (PAR) was provided for plants in each model unit on a day/night cycle of 14/10 hours by a *Camplex Plantcare* 160W Mercury Fluorescent Plant Irradiator, Model no. HD71026M (Thermoforce Limited). Each 'Irradiator' was positioned 90 cm above the centre of the soilbed surface downslope of each pole section (section 3.2.2.4.).

3.2.3.3. Temperature and relative humidity.

Temperature and relative humidity, measured at the soil surface using a *Vaisala* HM 34 Humidity and Temperature Meter, fluctuated between 19 and 23^{0}C and 50 and 60 %.

3.2.3.4. Soil moisture content.

The topsoil water content of each soil bed was monitored daily using a *Camplex Plantcare* soil moisture meter and probe, Model no. HD500M (Bentall Simplex Limited), previously calibrated to the percentage water holding capacity of soil. Field capacity was maintained by watering with tapwater using a hand held 1 dm^3 capacity mist sprayer.

3.2.4. Model operation and sampling.

3.2.4.1. Introduction.

Ten days after Rentex treated and control pole sections were positioned in each soil bed (section 3.2.2.4.), seed of the perennial ryegrass variety 'Fennema' (section 3.2.3.1.) was hand sown to each soil surface at a rate of 90 g/m^2. This heavy seeding rate was designed to ensure a uniform growth of grass in each model unit thereby eliminating plant density as an experimental variable.

Following the establishment of a uniform emergence of seedlings, 5 days after sowing, the first application of simulated rainfall was made to each pole section and soil bed. Table 3.2.2 shows the entire experimental schedule from the positioning of pole sections onwards. This table gives the times of simulated rainfall applications, leachate collections, plant sowings and the times of plant, soil and pole section sampling prior to more detailed description.

3.2.4.2. Simulated rainfall application.

Annual rainfall records, from 1980 - 90, for a Rentex field site at Glenclova (sections 2.2.3., 2.2.4. and 2.2.5.), were obtained from the Tay River Purification Board (Perth). The records indicated a mean annual rainfall of 1246 mm at this site, which, multiplied by the 1 m^2 coverage of each spray head (section 3.2.2.5.1.), would be achieved for each model unit by applying a total of 1246 dm^3 of tapwater through each spray head. Nine applications of simulated rainfall were made to each model unit in 40 days (table 3.2.2), each spray head providing a total volume of approximately 623 dm^3, equivalent to half of the mean annual rainfall figure for the Glenclova site. The volume of each simulated rainfall application was controlled by precise timing of spray head operation based on the measured flow rate

through each spray head of 0.368 dm^3min^{-1} (section 3.2.2.5.2.). Each of the first 3 rainfall simulations was carried out for 141 minutes to apply approximately 52 dm^3 of tapwater through each spray head at each simulation. To provide a more severe leaching environment within each model unit, rainfall simulations 4 - 9 were each carried out for 212 minutes to apply approximately 78 dm^3 of tapwater. As the surface area of each soil bed was 59.4 % of the coverage of each spray head (section 3.2.2.5.1.) the volume of tapwater entering each soilbed due to rainfall applications of 52 and 78 dm^3 was 30.89 and 46.33 dm^3 respectively, giving a total volume for 9 applications of approximately 370.65 dm^3.

Table 3.2.2. Schedule of the main practical procedures carried out over the period of the experiment.

Day of Experiment	Procedure
1	Pole Sections positioned in Soilbeds.
10	1st Sowing of Perennial Ryegrass.
15	1st Simulated Rainfall Application (SRA) and Leachate Collection (LC).
19	2nd SRA and LC.
22	3rd SRA and LC.
25/26	1st Sown Perennial Ryegrass Sampled.
28	4th SRA and LC.
32	5th SRA and LC.
37	6th SRA and LC.
40	2nd Sowing of Perennial Ryegrass.
47	7th SRA and LC.
52	8th SRA and LC.
55	9th SRA and LC.
58 - 61	2nd Sown Perennial Ryegrass Sampled.
104	Sowing of Rye.
126	Rye Seedling Emergence Count.
176 - 178	Rye Sampled.
186/187	Soilbeds Sampled, Water Tables Drained.
190	Pole Sections Removed and Sampled.

3.2.4.3. Leachate collection and sampling.

On commencement of the first rainfall simulation on day 15 (table 3.2.2), the ports of drains 5 and 9 of each model unit (figure 3.2.5.) were lowered to allow drainage of the raised watertable (section 3.2.2.4.). This drainage water was collected, together with the volumes of leachate moving through the drains of each soil bed, in the containers provided (section 3.2.2.2.). Twenty four hours after the first rainfall simulation was stopped the flow of leachate from all operative drains, except 5 and 9, had ceased. To facilitate the maintenance of field capacity in each soil bed, these latter drains were again inverted, allowing approximately 10 cm of leachate to accumulate in the base of each model unit. These procedures were followed for all 9 leachate collections (table 3.2.2).

The leachate from all three soilbeds was of a clear yellow colour. The leachate collected in each container after each rainfall simulation was thoroughly agitated and a known volume, usually 300 cm^3, was retained in a polyethylene screw top bottle for determination of pH and fluoride, total chromium and chromium (VI) content (section 3.2.5.1.). The volume of remaining leachate was carefully measured using 100 cm^3 and 1 dm^3 capacity measuring cylinders, then discarded, and the total volume of leachate in each container, including that portion retained for chemical analysis, was recorded.

To determine the same chemical characteristics for tapwater before its entry into the model units, 300 cm^3 of the simulated rainfall collected via gutters between the model units (section 3.2.2.2.) was similarly retained after each rainfall simulation and the remainder discarded. On completion of the final leachate collection the watertable in each model unit was increased to a depth of 20 cm once more (section 3.2.2.4.) and maintained at this level (section 3.2.3.4.) till each unit was drained on day 186/187 of the experiment (table 3.2.2). The volumes of these leachates were recorded as before and a portion retained for chemical analysis (section 3.2.5.1.).

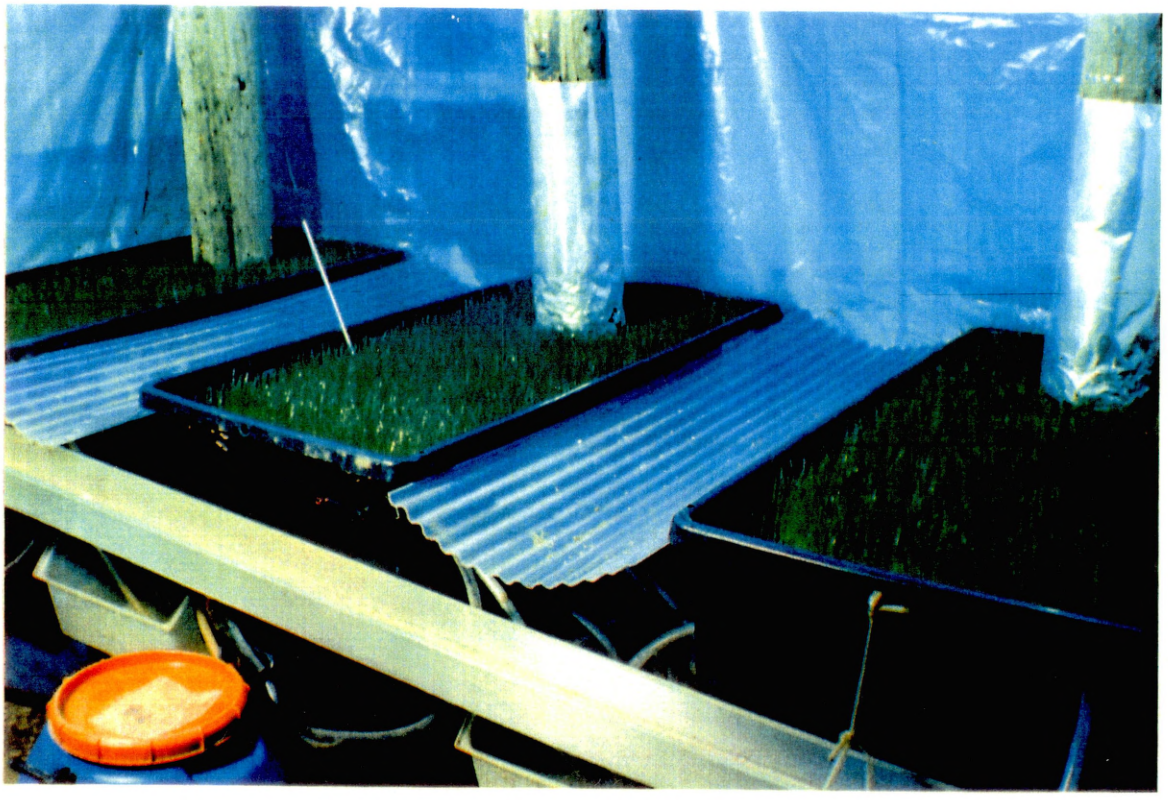
3.2.4.4. Seeding and sampling of perennial ryegrass.

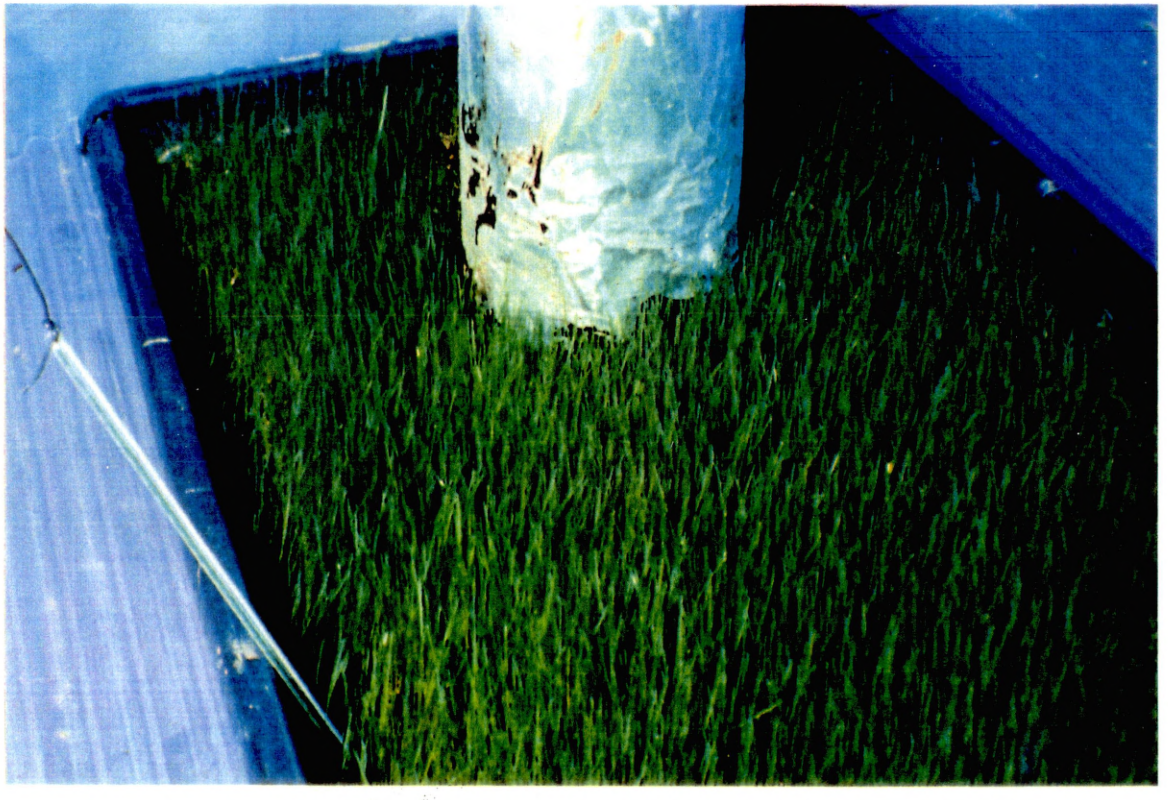
3.2.4.4.1. Seeding and sampling of first grass sward.

Seed of the perennial ryegrass variety 'Fennema' was handsown to each soil bed, at a rate of 90 g/m^2 on day 10 of the experiment (table 3.2.2) and uniform grass swards were quickly established (Plates 3.1 and 3.2). However, on days 23 and 24 of the experiment a slight general chlorosis was noted in the leaves of all 3 swards. Before further deterioration could take place, each sward was sampled on days 25 and 26 of the experiment (table 3.2.2) when the plants were approximately 10 cm in height.

The sward on each soilbed was sampled according to a sampling plan (figure 3.2.6). Samples from areas A - I (each measuring 25 cm^2), up to 15 cm downslope of each pole section (figure 3.2.6), were cut with stainless steel dissecting scissors. To provide uniformity of sampling, a square sided aluminium sampling tool with sides 2.5 cm deep and 5 cm long was carefully adjusted into position and only the grass above the 2.5 cm side of this sampling device was retained. Sward samples were similarly removed from areas J - L (each measuring 50 cm^2) situated between 25 - 35 cm downslope of each pole section (figure 3.2.6). For each soilbed, the plant material recovered from symmetrically opposite labelled sectors A - L (figure 3.2.6) was combined and retained in closed paper bags. A further single sample, consisting of all the grass from the remaining unsampled area within 5 cm around each pole section, was similarly cut and retained. This procedure provided 13 sward samples from each soilbed.

The remaining grass outside these sampled areas was cut to an approximate height of 2.5 cm and a representative sample for each soil bed was obtained by spreading and quartering (section 2.2.6.4.) and similarly retained.





Plates 3.1 and 3.2. Uniform swards of perennial ryegrass (***L. perenne***) established on the soil beds of the three model units approximately one week after sowing.

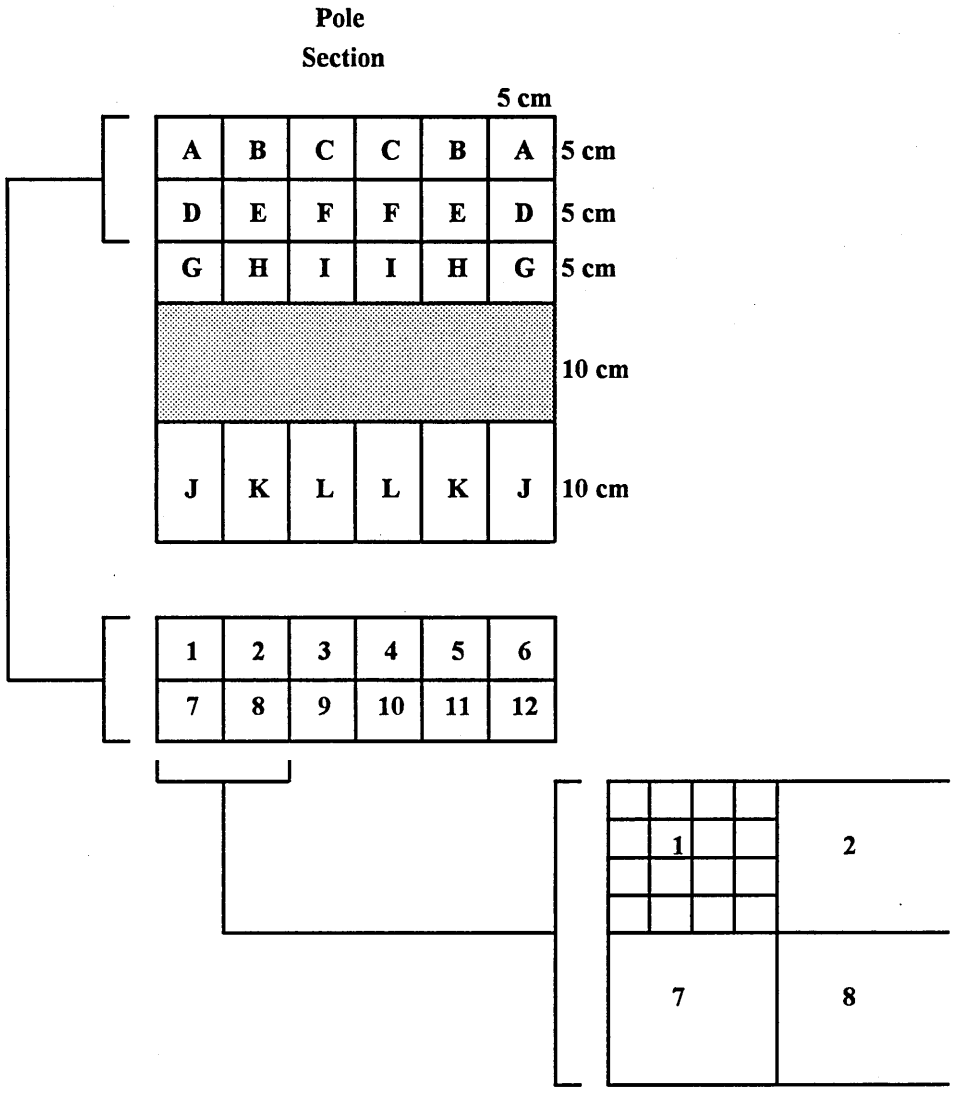


Figure 3.2.6. Diagram of perennial ryegrass sampling plan/grid positioned immediately downslope of the pole section in each model unit, showing labelled sampling areas A - L. Sectors 1 - 12 represent the galvanised mesh grid applied to each soilbed surface for sampling of the second ryegrass sward.

At the end of each day's sampling, the samples were removed from the bags and washed by prolonged spraying with distilled water. The samples were left to drain briefly on absorbent paper then placed in labelled clean paper bags. The bags were left slightly open in the laboratory for 1 week at an ambient temperature of approximately 21°C to facilitate air drying of the samples. The bags were sealed and samples retained for dry weight measurement (section 3.2.5.2.1.) and for chemical analysis of their fluoride and chromium content (section 3.2.5.2.2.).

Though this sampling procedure caused substantial disturbance and some minor sward damage, the freshly cut swards were considered to be in good condition. However, despite an initial indication of new growth up to 3 days after the grass was sampled, this was short-lived and necrosis of the remaining plant material occurred rapidly. Thereafter, all dead plant material was removed from the surface of each soilbed by hand.

3.2.4.4.2. Seeding and sampling of second grass sward.

A second seeding of ryegrass on day 40 (table 3.2.2) was necessitated by the failure of the first sward after sampling (section 3.2.4.4.1.). To reduce sward disturbance on sampling while retaining the original sampling plan (section 3.2.4.4.1.), a rigid mesh of galvanised steel, consisting of 25 cm^2 sectors each containing a further 16 smaller square sectors, was applied to the surface of each soil bed prior to seeding to form a fixed sampling grid (figure 3.2.6). The mesh was measured and cut to overlap the original sampling area downslope of each pole section. The overlaps were folded at right angles to the mesh face thereby supporting it at 2.5 cm above the soil surface and maintaining the original cutting height (section 3.2.4.4.1.). Sowing of the second sward at the same seeding rate as before (section 3.2.4.4.1.) was not restricted by the presence of the mesh sampling grid.

On days 58 - 61 of the experiment, as the sward was cut, the number of grass leaves growing through each of the 16 square sectors within each grid section labelled 1 to 12, in

each soilbed (figure 3.2.6), was recorded. For each soilbed; bulking of cut samples from grid sections A to L (figure 3.2.6); sampling of grass around the pole section; sampling of remaining uncut grass and the treatment of all samples recovered was as described for the first grass swards (section 3.2.4.4.1.).

As before, up to 3 weeks after the second grass swards were sampled no indications of new growth were evident. Thereafter a progressive necrosis of the plants resulted in the death of the second swards approximately 2 weeks later, and all dead plant material was removed by hand from each soilbed.

3.2.4.5. Seeding, sampling and measurement of rye plants.

3.2.4.5.1. Seeding.

On day 104 of the experiment, rye seed (section 3.2.3.1.) was sown in each grass free soilbed (table 3.2.2) at the seed suppliers recommended seeding rate of 645/m^2 or 383 seeds per soilbed. The seeds were planted in staggered rows at a recommended depth of 2.5 cm, using a 0.5 cm diameter dibble. The 1st seed was planted centrally 1 cm downslope of each pole section. The distance between seed rows and between planted seeds within each row was 4 cm.

3.2.4.5.2. Maintenance of erect plant growth habit.

Approximately 2 weeks after sowing (section 3.2.4.5.2.), the development of a reclining growth habit was noted in the majority of rye seedlings in each soilbed (section 3.2.3.1.). Though there were no indications of damage or disease in these plants, physical supports were used to re-establish an erect growth habit to prevent possible rotting of leaves in contact with the moist soil, and/or disruption of translocation within the stems. The

supports consisted of short lengths of 2 mm bore glass tubing inserted into the soil adjacent to each seedling. Lengths of plastic coated fuse wire were threaded into the open upper ends of the tubes, leaving a wire protruding from each, which was loosely wound round each plant.

3.2.4.5.3. Seedling count.

On day 126 of the experiment, approximately 1 week after rye seedling supports were placed in each soilbed (section 3.2.4.5.2.), a count of seedlings was carried out to determine the number of viable plants reaching this seedling stage in each soilbed. 'Seedlings' described as non-viable at the time of the count were those which had not emerged after sowing or had died in the intervening period. The count was restricted to 4 seed rows either side of each pole section, numbered 1-4, and 14 rows immediately downslope of each pole section, numbered 5-18 , representing a total of 179 planted seeds (figure 3.2.7).

3.2.4.5.4. Plant community development.

A progressive reduction in plant density within the rye plant community downslope of each pole section, including the control, was noted during the weeks following the viable seedling count (section 3.2.4.5.3.). As expected, this reduction was accompanied by an increase in the size of individual rye plants in this area. A more pronounced reduction in plant density occurred in the soilbed to the sides and rear of each pole section over the same period of time. Irrespective of pole section treatment (section 3.2.2.4.), the few surviving plants in this area were commonly stunted and most consisted of largely necrotic tissue. These symptoms may have been due to the cumulative effect of the very moist soil conditions, maintained by watering (section 3.2.3.4.) and a raised watertable (section 3.2.4.3.), allied to the greater shading of these plants, resulting from the centralisation of the lighting units above the soilbed area downslope of each pole section (section 3.2.3.2.). Consequently, these plants were removed and discarded prior to sampling of the rye plants

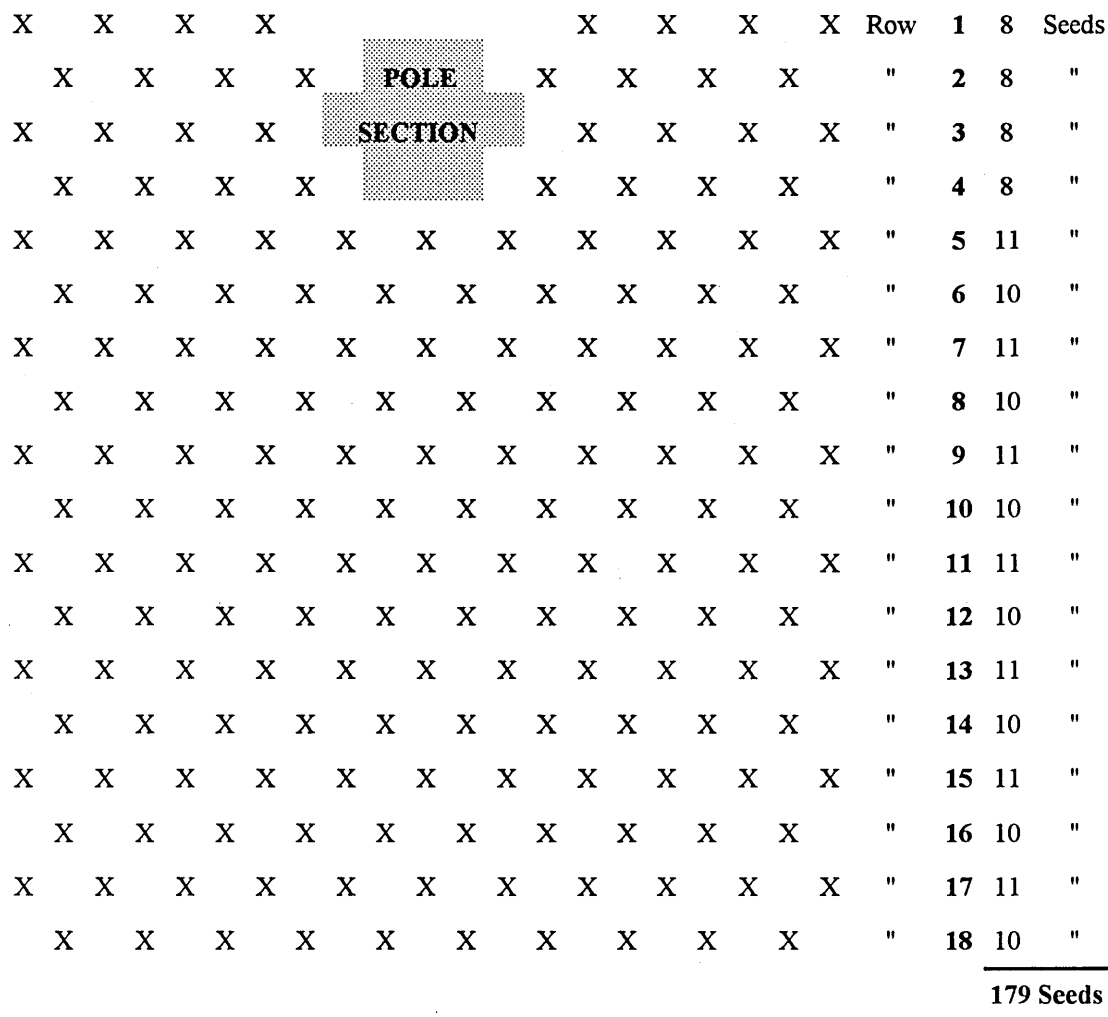


Figure 3.2.7. Plan of staggered seed rows used to count viable Rye seedlings in each soilbed on day 126 of the experiment approximately 3 weeks after sowing.

downslope of each pole section (section 3.2.4.5.5.).

3.2.4.5.5. Plant sampling and measurement.

Sampling of plant material from the area downslope of the pole section in each soilbed was carried out on days 176 - 178 of the experiment, over 2 months after seeding (table 3.2.2). The sampling time was dictated by the necessity to retrieve a sufficient number of individual plants for meaningful statistical comparisons of growth parameters.

To reduce disturbance and damage to the rye plant canopy, which occurred when whole plant removal was initially attempted, roots were collected after recovery of all shoots from each soilbed. To allow reasonable comparison of plant measurements within and between soilbeds, individual plants were assigned to specific soilbed sectors, each measuring 60 cm x 10 cm, within six 10 cm increments downslope of each pole section (figure 3.2.8).

The height of each plant's highest part within the canopy, and each plant's position relative to the pole section, measured at the plants groundline, were recorded. The shoot was cut at the groundline using stainless steel dissecting scissors and lifted free of the surrounding plant material. The position of the roots was identified with a black plastic marker placed on the soil surface. The number and total length of viable leaves was immediately recorded for each plant, as was the number of non-viable leaves. Viable and non-viable leaves were those displaying less than or more than 50 % chlorosis of the leaf length respectively. Each entire shoot was retained separately in a labelled paper sample bag.

When all shoots had been recovered from each soilbed, the soil containing each marked root system was gently uplifted using a 15 cm long aluminium dissection probe. The soil was lightly shaken off the roots, into its original position in the soilbed, and the roots were gently washed with distilled water to remove any excess soil. Due to the abrasive nature of

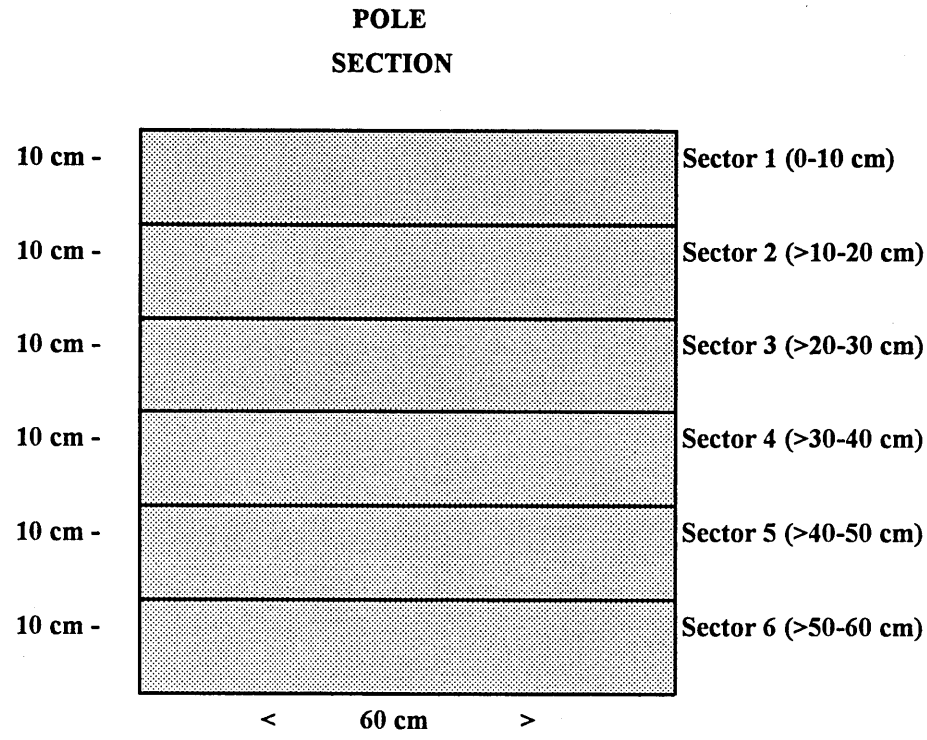


Figure 3.2.8. Diagram (not to scale) of the soilbed surface downslope of each pole section showing 6 sampling areas, each approximately 600 cm , to which individual Rye plants were assigned. This sampling plan provided a basis for statistical comparisons of Rye plants within and between model units.

the soilbeds this procedure was successful in providing only a limited number of complete root systems. Examination of the roots under low magnification, immediately after washing, indicated no apparent anatomical differences between root systems recovered from different soilbeds. The moist roots were placed on absorbent paper to drain, and for each root system the length of the longest root axis was recorded. Each root system was retained separately in a labelled paper sample bag. All paper sample bags, containing shoot and root samples, were placed overnight in an oven set at 105°C, and the dry weight of each shoot and each root system was recorded separately.

3.2.4.6. Soilbed sampling.

On days 186 and 187 of the experiment, approximately 1 week after rye plants were sampled (section 3.2.4.5.5.), soil samples were extracted from each soilbed (table 3.2.2) using a long handled stainless steel scoop. For each soilbed containing a Rentex treated pole section (section 3.2.2.4.), 2 soil cores, each measuring 5 cm x 5 cm x 15 cm deep and weighing approximately 1 kg, were extracted at 0 - 15 cm and 15 - 30 cm depths from the top 30 cm of soil at each of the numbered positions 1 - 6 shown in figure 3.2.9. Soil samples from each depth at like numbered positions were bulked and homogenised giving 6 soil samples from each depth for each soilbed containing a Rentex treated pole section. Soil samples were similarly recovered from positions 1, 3, 5 and 6 (figure 3.2.9) of the soilbed containing the untreated pole section. Soil samples 1 and 3 (figure 3.2.9) were bulked for each depth, thereby providing 3 soil samples from each depth for this soilbed.

In addition, representative watertable mud samples from each soilbed were recovered between a depth of 30 cm and the base of each soilbed, from the same sampling positions, excepting position 2 of each soilbed containing a Rentex treated pole section. For all 3 soilbeds, the samples recovered at this depth from positions 1 and 3 (figure 3.2.9) were bulked and homogenised. This provided a further 4 samples for each soilbed containing a

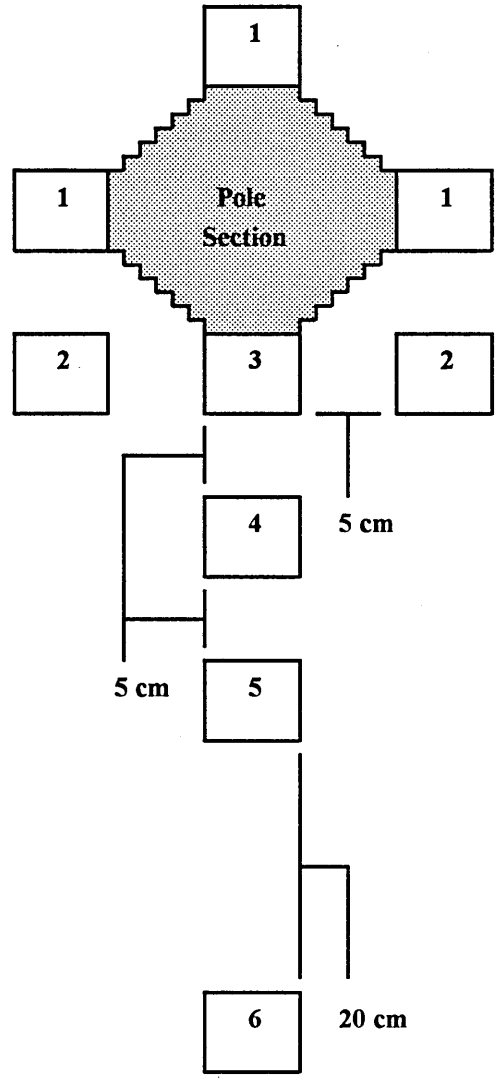


Figure 3.2.9. Sampling plan, not to scale, for removal of soil cores 1 - 6 from the soilbed of each model unit. Each corehole measured 5 cm x 5 cm at the soil surface.

Rentex treated pole section and 3 for the soilbed containing an untreated pole section.

Representative sub-samples of each moist soil sample recovered were obtained by spreading and quartering (section 2.2.6.4.) and placed in plastic sample bags for determination of fluoride and chromium contents (section 3.2.5.4.1.). The remainder of each soil sample recovered from the 2 lower depths in each soilbed was discarded. The remaining bulk of each soil sample recovered from the top 15 cm of each soilbed was retained for measurements of soil dehydrogenase activity (section 3.2.5.4.2.).

3.2.4.7. Sampling of Rentex treated pole sections.

On day 190 of the experiment, the pole sections were removed from the soilbeds (table 3.2.2). Two sets of wood samples were recovered from each of the 2 Rentex treated pole sections removed from 2 soilbeds and the 2 Rentex treated pole sections mantained under the same conditions of temperature and relative humidity (section 3.2.3.3.) outside the soilbeds (section 3.2.2.4.).

Sawdust samples from each pole section were produced by sectioning the treated zone of each pole section, as for those sawdust samples from field pole sections (section 2.2.6.4.). This procedure provided 3 sawdust samples from each Rentex treated pole section for determination of fluoride and chromium content (section 3.2.5.5.) to identify any leaching loss of these preservative constituents from those pole sections removed from the soilbeds.

3.2.5. Analyses of samples recovered from the model system.

3.2.5.1. Chemical analysis of simulated rainfall and soil leachate.

3.2.5.1.1. 'Rainfall' and leachate sample preparation.

The 'rainfall', and soilbed leachate samples (section 3.2.4.3.) retained after each of the 9 simulated rainfall applications (section 3.2.4.2.) were apparently clear of suspended soil particles. However, to prevent blockage of the plastic capillary sampler of the atomic absorption spectrophotometer during total chromium measurements (section 3.2.5.1.4.), each sample was clarified further by filtering a portion for analysis through Whatman No. 4 filter paper. These filtered samples, 45 leachates from the 5 separate drains of each model unit containing a Rentex treated pole section, 18 leachates from the 2 combined drains of the control model unit and 9 samples of 'rainfall', were used for all analytical measurements, except where indicated otherwise. The samples of leachate collected from the model units when each was drained at the end of the experiment (section 3.2.4.3.), were similarly filtered. These latter samples were analysed for chromium (VI) content (section 3.2.5.1.5.). Analar quality reagents and grade A glassware were used for all analyses.

3.2.5.1.2. Measurement of pH in 'rainfall' and leachate samples.

The sample bottles containing unfiltered leachate or 'rainfall' samples (section 3.2.4.3.) were thoroughly agitated and a single measurement of pH was carried out for each using a Corning Eel Model 12 pH meter and electrode previously calibrated with buffer solutions of pH 4.6, 7.0 and 9.4.

3.2.5.1.3. Determination of fluoride concentration in 'rainfall' and leachate samples.

For each filtered leachate and 'rainfall' sample (section 3.2.5.1.1.), duplicate measurements of the fluoride concentration (ug/cm^3) were carried out using the selective ion electrode method (section 2.2.2.3.4.), which was checked for accuracy by determining the fluoride content of aqueous samples of variable pH (appendix 2). The fluoride concentration was multiplied by the total volume of each original leachate solution (section 3.2.4.3.) or by the calculated volume of 'rainfall' entering each model unit at each simulation (section 3.2.4.2.) to give total quantities of fluoride in ug and mg.

3.2.5.1.4. Determination of total chromium concentration in 'rainfall' and leachate samples.

Summary.

For each filtered 'rainfall' and leachate sample (section 3.2.5.1.1.), duplicate measurements of total chromium concentration (ug/cm^3) were carried out using the atomic absorption addition calibration method (section 2.2.2.3.5.), which was checked for accuracy by determining the total chromium content of standard aqueous samples of variable pH (appendix 3).

Simulated rainfall samples.

For measurement of total chromium in each filtered 'rainfall' sample (section 3.2.5.1.1.), addition calibration solutions (section 2.2.2.3.5.) were prepared in four volumetric flasks of 25 cm^3. Flask 1 was made up to volume with distilled water as a blank solution and flasks 2, 3 and 4 were made up to volume with representative portions of the 1st 'rainfall' sample collected (section 3.2.4.3.). Using a positive displacement micropipette, 50 ul of a 25 ugCr/cm^3 solution and 25 ul of a 100 ugCr/cm^3 prepared by dilution of a standard

chromium solution (section 2.2.2.3.2.), were added to flasks 3 and 4 respectively. The chromium concentrations of solutions in flasks 3 and 4 were therefore equivalent to 0.05 and 0.1 ugCr/cm^3 respectively, in addition to the the unknown sample concentration of chromium in flask 2. A further 17 volumetric flasks of 25 cm^3 were prepared and numbered 5 to 21. Flask 5 was made up to volume with a representative portion of the 1st 'rainfall' sample collected (section 3.2.4.3.), ie. a duplicate of flask 2. Flasks 6/7, 8/9, 10/11, 12/13, 14/15, 16/17, 18/19 and 20/21 were made up to volume with representative portions of the 2nd - 9th 'rainfall' samples respectively.

The absorbances of the 21 solutions were read and the chromium concentration of each duplicated 'rainfall' sample solution was displayed in ug/cm^3 (section 2.2.2.3.5.). Between readings of each duplicated 'rainfall' sample, the 0.05 ugCr/cm^3 solution in flask 3 was read to verify the accuracy of measurements, and indicated a mean total concentration of 0.05 ugCr/cm^3 with a standard deviation of 0.009 ugCr/cm^3. The mean total chromium concentration, in ug per cm^3, for each duplicated 'rainfall' sample was multiplied by the calculated volume of 'rainfall' entering each model unit at each rainfall simulation to give total quantities of chromium, entering each model unit at each rainfall simulation, in ug and mg.

Soil leachate samples.

Each leachate sample was allocated to a group of 9 corresponding to the 9 samples collected from any one particular drain after the 9 simulated rainfall events (section 3.2.4.3.) for each of the 5 operative drains (section 3.2.2.2.) of each of the model units containing a Rentex treated pole section (section 3.2.2.4.) or to a group of 9 leachate samples (section 3.2.4.3.) for each of the 2 combined drains (section 3.2.2.2.) of the model unit containing an untreated pole section (section 3.2.2.4.). This provided 12 separate 'drain' groups of 9 leachate samples. Separate addition calibration solutions (section 2.2.2.3.5.) were prepared for each 'drain' group, as for 'rainfall' samples, from the 1st leachate sample collected

(section 3.2.4.3.) for each 'drain' group. However, since soil leachates were likely to contain a greater range of chromium concentrations in excess of 'rainfall' samples, the number and concentration of addition calibration solutions was increased (table 3.2.3).

Table 3.2.3. Addition calibration solutions for measurement of total chromium concentration in each filtered leachate sample of each 'drain' group.

	Flask.(25 cm^3)		Total Cr.(ug/cm^3)
1 -	No Cr addition. Distilled water blank.	=>	0.00
2 -	No Cr addition. 1st leachate sample.	=>	Unknown
3 -	1st leachate sample + Cr addition	=>	0.05 + Unknown
4 -	As above	=>	0.10 + "
5 -	As above	=>	0.50 + "
6 -	As above	=>	1.00 + "
7 -	As above	=>	2.50 + "

The total chromium concentration in each filtered leachate sample (section 3.2.5.1.1.) of each group was determined in duplicate, as for 'rainfall' samples.

3.2.5.1.5. Determination of chromium (VI) concentration in 'rainfall' and leachate samples.

Summary.

Determination of chromium (VI) concentration in each 'rainfall' and leachate sample was carried out using an adapted spectrophotometric method (Charlot, 1964). Two determinations were carried out for each aqueous sample, the 1st after collection (section 3.2.4.3.) and the 2nd, approximately 18 weeks later. Both determinations were carried out on freshly filtered (section 3.2.5.1.1.) samples.

Apparatus.

Readings of each sample were taken with a Perkin Elmer UV/VIS Spectrophotometer Lambda 2.

Reagents.

Phosphoric acid solution (PA) consisted of concentrated phosphoric acid (40 cm^3) made up to 1 dm^3 in distilled water. Diphenylcarbazide solution (DPC) consisted of diphenylcarbazide (0.25 g) dissolved and made up to 100 cm^3 in acetone. A chromium (VI) standard solution (1000 ugCr/cm^3) consisted of potassium dichromate (2.8290 g) dissolved and made up to 1 dm^3 in distilled water.

Method.

Measurement.

The concentration of chromium (VI) in ug/cm^3 for each 'rainfall' and leachate sample solution was measured at a wavelength of 540 nm by reference to a linear standard curve of solutions containing known concentrations of chromium (VI).

Calibration solutions.

To accomodate the variability in chromium (VI) concentrations found in the samples, 2 ranges of standard curve solutions were prepared. Separate additions of 0, 0.5, 1, 3 and 5 cm^3 of a chromium (VI) solution (5 ug/cm^3), prepared by dilution of the chromium (VI) standard solution, were made to 5 volumetric flasks (25 cm^3), each containing PA (10 cm^3) and DPC (2.5 cm^3). The solutions were made up to volume with distilled water to provide chromium (VI) concentrations of 0, 0.1, 0.2, 0.6 and 1.0 ug/cm^3. Standard curve solutions

of 0.01, 0.02, 0.06 and 0.10 ug/cm^3, for measurement of smaller concentrations of chromium (VI), were provided by adding 2.5 cm^3 of a 0.1, 0.2, 0.6 and 1.0 ug/cm^3 chromium (VI) solution, separately, to similarly prepared flasks.

Sample solutions.

Aliquots (5-10 cm^3) of each filtered 'rainfall and leachate sample were added to each of 2 volumetric flasks (25 cm^3). PA (10 cm^3) and DPC (2.5 cm^3) were added to each flask which were made up to volume immediately with distilled water. Each group of samples for measurement was accompanied by a blank solution, consisting of PA (10 cm^3) and DPC (2.5 cm^3) made up to volume with distilled water, for measurement correction. The flasks were left to stand for 10 minutes before measurement to allow development of a deep red colour indicative of the presence of chromium (VI). The accuracy of chromium (VI) measurements of these samples was confirmed periodically by measurements of similarly prepared leachate solutions containing additions of chromium (VI) equivalent to concentrations of 0.20 and 0.02 ug/cm^3.

Calculation of chromium (VI) concentration.

Calculation of chromium (VI) concentration in each aqueous sample was carried out as indicated for the following hypothetical solution, consisting of a 5 cm^3 aliquot of a leachate sample in a 25 cm^3 flask, giving a concentration reading of 0.8 ug/cm^3 chromium (VI):

Concentration (ug/cm^3) x Volume of flask (cm^3) = No. of ug present in flask

=> 0.8 x 25 = 20 ug

Therefore, the 5 cm^3 aliquot sample contained 20 ug and the chromium (VI) concentration of the original leachate sample was 4 ug/cm^3.

3.2.5.2. Measurement and chemical analysis of ryegrass samples.

3.2.5.2.1. Dry weight measurement of ryegrass samples.

For each ryegrass sward of each model unit (sections 3.2.4.4.1. and 3.2.4.4.2.), air dry weights were recorded for the 12 combined grass samples A to L (figure 3.2.6.); the single sample from around the pole section; and 2 sub-samples of the grass recovered outside these areas. The 6 pairs of sub-samples (1 pair for each of the 2 swards grown on each soilbed) were placed overnight in an oven set at 105°C. Oven dry weights were recorded and the sub-samples discarded. The oven dry weights of each pair of sub-samples were used to calculate a correction factor to determine the dry weights of the remaining 13 air dried samples from their respective swards as follows:

$$\frac{\dfrac{\text{Oven dry weight 1 (g)}}{\text{Air dry weight 1 (g)}} + \dfrac{\text{Oven dry weight 2 (g)}}{\text{Air dry weight 2 (g)}}}{2} = \mathbf{A}$$

and **A** x Air dry weight of sample (g) = Dry weight (g).

3.2.5.2.2. Determination of fluoride and chromium content of ryegrass samples.

Without further preparation, each of the 78 entire grass samples (13 from each of the 2 swards grown on each soilbed) for which a dry weight was calculated (section 3.2.5.2.1.), was used for a single analysis for fluoride and chromium content in ug/g of dry weight using the same method as for the wood samples (section 2.2.2.3.).

3.2.5.3. Measurement of rye plant samples.

The observations and physical measurements of rye plants, carried out during plant establishment and sampling (sections 3.2.4.5.3. and 3.2.4.5.5. respectively) are detailed in these sections.

3.2.5.4. Chemical analysis of soilbed samples.

3.2.5.4.1. Determination of fluoride and chromium content of soilbed samples.

Representative sub-samples of each moist soil sample recovered from each soilbed (section 3.2.4.6.) were prepared and analysed for fluoride and chromium content as for field soils (section 2.2.5.4.).

3.2.5.4.2. Measurement of dehydrogenase activity in surface soil samples.

Introduction.

Measurements of enzyme inactivation are regarded as one of the most relevant techniques for determining harmful effects of pollutants on soil microflora (US EPA, 1978; Forstner, 1988) and many workers have measured soil dehydrogenase activity to assess potential detrimental effects of pesticides and heavy metals (Davies and Greaves, 1981; Mowe, 1983; Green, 1988; Hainey, 1992; Chander and Brookes, 1991 a). Therefore, after sub-samples of each moist soil sample, recovered from the top 15 cm of each soilbed, had been obtained (section 3.2.4.6.) for determination of fluoride and chromium content (section 3.2.5.4.1.) the remainder of each of these 15 soil samples was used for measurements of dehydrogenase activity. The standard procedure for measurement of soil dehydrogenase activity is via the reduction of 2,3,5 - triphenyltetrazolium chloride (TTC) to

the red coloured precipitate triphenylformazan (TPF). In the absence of 0_2, the usual terminal electron acceptor in the transfer of electrons carried out by the endocellular dehydrogenase enzymes, TTC acts as the terminal electron acceptor. Measurements were carried out using the spectrophotometric method of Casida *et al* (1964) as modified by Mowe (1983).

Apparatus.

Readings of each sample were taken with a Perkin Elmer UV/VIS Spectrophotometer Lambda 2.

Reagents.

Calcium carbonate. Ethanol. Triphenyltetrazolium chloride solution (0.75 w/v), TTC, consisted of 2, 3, 5 - triphenyltetrazolium chloride (1.8750 g) dissolved and made up to 250 cm^3 with distilled water. A triphenyltetrazolium formazan standard solution (0.333 umol/cm^3), TTF, consisted of 2, 3, 5 - triphenyltetrazolium formazan (0.0100 g) dissolved and made up to 100 cm^3 with ethanol.

Method.

Measurement.

The level of dehydrogenase activity in umol TTF/cm^3 for each soil sample solution was measured at a wavelength of 485 nm by reference to a standard curve of solutions containing known concentrations of TTF.

Calibration solutions.

TTF standard curve solutions containing 0.111, 0.055, 0.028, 0.014, 0.007 and 0.003 umol TTF/cm^3 were prepared by sequential dilutions of the standard TTF solution (0.333 umol TTF/cm^3) in ethanol.

Soil samples.

Each moist soil sample was seived through a 2 mm stainless steel sieve and placed in an unsealed plastic bag. The bags were placed together in plastic boxes, containing a 3 cm deep layer of water, which were loosely sealed. The boxes were left for 1 week, at a mean temperature and relative humidity of 20°C and 75 % respectively, to allow soils to reach equilibrium moisture content.

A representative 200 g portion of each of the 15 soil samples was adjusted to 20 % w/w moisture content by addition of distilled water and the remainder of each sample was discarded. Each sample was split to provide 2 portions of 100 g. One portion was supplemented by thorough mixing with 1 g of milled rye meal previously held at 120°C for 24 hours. The unsupplemented and supplemented samples were each split into 4 sub-samples of approximately 25 g giving a total of 120. Each sub-sample was placed into a loosely stoppered 70 cm^3 glass sample bottle to a depth of 2 cm. The bottles were stored for 4 weeks, in a covered ventilated plastic tray containing a 3 cm deep water layer, at a mean temperature and relative humidity of 18°C and 85 % respectively.

Soil sample solutions.

At 18 hours (representing 0 weeks), 1, 3 and 4 weeks after supplementation of soils, 15 pairs of bottles, 1 containing a supplemented sub-sample and the other not, were removed from the tray. Four 1.5 g sub-samples were removed from each bottle, 3 for dehydrogenase

activity measurement and 1 for dry weight correction (section 2.2.5.). The 3 replicates were each weighed into a separate screw top test tube (120 mm x 15 mm) containing calcium carbonate (15 mg) and 2 cm^3 of TTC. The contents of each test tube were thoroughly mixed on a vortex shaker and each tube was sealed and incubated at 30°C for 24 hours in darkness. After incubation, 5 cm^3 of ethanol was pipetted into each test tube and mixed for 5 minutes on a vortex shaker. On settling, the supernatant liquid in each test tube was decanted into a centrifuge tube. The remaining soil particles in each test tube were rinsed with a further 3cm^3 of ethanol, the supernatant was decanted again and the test tube contents discarded. The total decanted liquid was centrifuged (x 4000 g) for 5 minutes to separate any remaining soil particles.

Calculation of dehydrogenase activity.

The level of dehydrogenase activity in each soil sample expressed in umols TTF $g^{-1}min^{-1}$ was calculated using the equation,

$$\text{umol TTF } g^{-1}min^{-1} = \frac{\text{Sample solution concentration (umol TTF/cm}^3\text{)} \times 10}{\text{Dry weight of soil (g)} \times (24 \times 60)}$$

3.2.5.5. Determination of fluoride and chromium content of pole section wood samples.

Wood samples recovered from each Rentex treated pole section (section 3.2.4.7.) were analysed for fluoride and chromium content as for wood samples from field pole sections (section 2.2.2.3.).

3.3. RESULTS.

3.3.1. General layout and statistical treatment of results.

The results for the studies detailed in sections 3.2.4. and 3.2.5. are presented in tables at the beginning of sections 3.3.2., 3.3.3., 3.3.4., 3.3.5., 3.3.6. and 3.3.7., and are followed by descriptions of results. In these results sections, simulated rainfall and the model unit containing a non-remedially treated pole section are referred to as SR and CS respectively, while the model units TS and TSS are those which contained remedially treated pole sections, the latter in sand amended soil.

Oneway analysis of variance was employed for all statistical comparisons using the MINITAB statistical computer package (Copyright 1992 Minitab Inc.). Where oneway statistical comparisons of more than two values indicated significant differences, Scheffes S test for analysis of contrasts (Dowdy and Wearden, 1991) was employed to identify where these existed.

3.3.2. Simulated rainfall and soil leachate analyses.

3.3.2.1. Introduction.

Simulated rainfall applied to each model unit soil bed (section 3.2.4.2.) and the resultant leachates, from drains in each soilbed (section 3.2.2.2.), were collected. The quantity of leachate from each drain was measured for each model unit and samples of simulated rainfall and leachates were retained (section 3.2.4.3.) for chemical analysis to determine pH and fluoride, total chromium and chromium (VI) concentrations (for methods, see sections 3.2.5.1.2., 3.2.5.1.3., 3.2.5.1.4. and 3.2.5.1.5. respectively). Samples of leachate retained from the watertables of each model unit soil profile when each was finally drained at the end of the model trial (section 3.2.4.3.) were analysed for chromium (VI) content only.

Statistical comparisons (section 3.3.1.) of pH in leachates of CS, TSS and TS were confined to those mean values highlighted in table 3.3.2.2, while statistical comparisons (section 3.3.1.) of the fluoride and total chromium concentrations in these leachates were confined to those values shown in tables 3.3.2.5 and 3.3.2.9 (see section 3.3.2.2.).

3.3.2.2. Contents of results tables.

Water volumes: - Table 3.3.2.1 shows the volume of applied simulated rainfall, SR, and the leachates produced from drains of model units CS, TSS and TS at each of the 9 simulated rainfall applications over a 40 day period. Total volumes over all 9 applications for SR, CS, TSS and TS are also shown.

pH: - Table 3.3.2.2 indicates the pH of SR and leachates produced from CS, TSS and TS at each rainfall application, the mean pH combined for all leachates from each model

Table 3.3.2.1. Volume of simulated rainfall (SR) entering each model unit (CS, TSS and TS) and the volume of leachate produced from numbered drains of each model unit at each rainfall simulation.

Model Unit	Drain No.	Volume (dm^3) of 'Rainfall' Applied/Leachate Produced at Day:									Total
		15	19	22	28	32	37	47	52	55	
SR	**Total**	30.888	30.888	30.888	46.332	46.332	46.332	46.332	46.332	46.332	370.656
CS	**6, 7, 8**	06.360	06.935	07.645	12.915	13.745	13.215	11.640	10.370	11.690	094.515
	9, 5	16.410	16.420	16.350	20.755	24.565	23.725	23.270	23.530	23.730	188.755
	Total	22.770	23.355	23.995	33.670	38.310	36.940	34.910	33.900	35.420	283.270
TSS	**5**	00.755	00.240	00.150	X	01.645	02.075	03.940	04.150	03.630	016.585
	6	00.120	00.075	X	X	X	X	00.015	X	00.140	000.350
	7	00.505	01.450	01.600	02.355	01.915	02.000	01.970	01.850	02.480	016.125
	8	00.880	00.780	00.135	00.465	00.057	00.590	00.250	X	00.075	003.237
	9	24.760	21.600	21.255	35.855	36.120	33.155	33.445	31.960	30.470	268.620
	Total	27.020	24.150	23.140	38.675	39.737	37.820	39.620	37.960	36.795	304.917
TS	**5**	00.700	00.445	00.110	09.555	03.865	08.230	06.930	10.450	06.030	046.315
	6	01.800	01.265	00.580	02.195	02.375	01.220	01.460	01.510	01.710	014.115
	7	02.725	02.900	02.735	02.955	03.035	03.420	03.330	04.210	03.270	028.580
	8	05.140	03.595	02.725	10.135	07.355	05.930	08.150	07.290	08.740	059.060
	9	17.340	16.415	17.605	16.575	25.145	18.500	18.210	15.150	19.840	164.780
	Total	27.705	24.620	23.755	41.415	41.775	37.300	38.080	38.610	39.590	312.850

X = Blocked Drain/No Leachate Collected.

unit at each rainfall simulation, and the mean pH of SR and each leachate of each model unit over all 9 rainfall simulations.

Fluoride: - Table 3.3.2.3 shows the mean fluoride concentrations respectively in SR and separate leachates from CS, TSS and TS at each rainfall application.

Fluoride: - Table 3.3.2.4 displays numerical and figurative expressions of the quantities of fluoride in SR and the total leachates from CS, TSS and TS at each rainfall simulation and numerical expressions of the total quantities of fluoride in SR and the total leachates from CS, TSS and TS over all rainfall simulations.

Fluoride: - Table 3.3.2.5 shows numerical and figurative expressions of the mean fluoride concentration in each numbered leachate from each model unit over all rainfall applications.

Fluoride: - Table 3.3.2.6 displays the concentrations of fluoride in each numbered leachate of CS, TSS and TS (table 3.3.2.3) multiplied by their respective volumes (table 3.3.2.1) and combined for each model unit to give numerical and figurative expressions of the total quantities of fluoride found in the leachates from separate drains of each model unit over all rainfall simulations. The total volume of each separate leachate is also shown.

Total Chromium: - Table 3.3.2.7 shows the mean total chromium concentrations respectively in SR and separate leachates from CS, TSS and TS at each rainfall application.

Total Chromium: - Table 3.3.2.8 displays numerical and figurative expressions of the quantities of total chromium in SR and the total leachates from CS, TSS and TS at each rainfall simulation (table 3.3.2.3) and numerical expressions of the total quantities of total chromium in SR and the total leachates from CS, TSS and TS over all rainfall simulations.

Table 3.3.2.2. The pH of simulated rainfall (SR) entering each model unit (CS, TSS and TS) and the pH of leachate produced from numbered drains of each model unit at each rainfall simulation (standard deviations in parenthesis).

Model Unit	Drain No.	pH of Simulated Rainfall Applied/Leachate Produced at Day:									**Mean**
		15	**19**	**22**	**28**	**32**	**37**	**47**	**52**	**55**	
SR		6.75	7.00	5.90	5.90	5.90	6.10	6.90	7.00	7.20	**6.52** (0.55)
CS	**6, 7, 8**	6.75	6.85	6.45	6.05	6.35	6.95	7.10	7.05	7.35	**6.77** (0.41)
	9, 5	7.05	5.25	5.40	5.70	5.75	5.80	7.05	6.80	7.00	**6.20** (0.76)
	Mean	**6.90**	**6.05**	**5.92**	**5.88**	**6.05**	**6.38**	**7.08**	**6.92**	**7.18**	**6.48** (0.66)
		(0.21)	(1.13)	(0.74)	(0.25)	(0.42)	(0.81)	(0.04)	(0.18)	(0.25)	
TSS	**5**	7.70	7.30	7.00	X	6.00	6.75	6.95	6.90	6.95	**6.94** (0.48)
	6	7.35	7.10	X	X	X	X	X	X	7.10	**7.18** (0.14)
	7	7.05	6.90	6.25	6.35	6.10	7.00	7.15	7.15	7.40	**6.82** (0.46)
	8	7.40	7.35	6.30	6.00	X	7.25	6.70	X	X	**6.83** (0.59)
	9	7.15	6.45	5.95	5.75	5.75	6.95	7.05	7.00	7.05	**6.57** (0.60)
	Mean	**7.33**	**7.02**	**6.38**	**6.03**	**5.95**	**6.99**	**6.96**	**7.02**	**7.12**	**6.82** (0.52)
		(0.25)	(0.36)	(0.44)	(0.30)	(0.18)	(0.21)	(0.19)	(0.13)	(0.19)	
TS	**5**	7.70	7.10	7.00	6.00	5.75	6.90	6.90	6.70	6.80	**6.76** (0.58)
	6	7.05	6.90	6.65	6.15	6.05	7.25	7.15	7.15	7.45	**6.87** (0.49)
	7	6.75	6.90	6.45	6.55	6.00	7.25	7.40	7.20	7.50	**6.89** (0.50)
	8	6.70	6.80	6.50	6.10	6.25	7.00	7.25	7.15	7.50	**6.81** (0.47)
	9	7.30	6.70	6.00	5.90	6.05	7.35	7.20	7.25	7.15	**6.77** (0.62)
	Mean	**7.10**	**6.88**	**6.52**	**6.14**	**6.02**	**7.15**	**7.18**	**7.09**	**7.28**	**6.82** (0.51)
		(0.41)	(0.15)	(0.36)	(0.25)	(0.18)	(0.19)	(0.18)	(0.22)	(0.31)	

X = Blocked Drain/No Leachate Collected.

Table 3.3.2.3. Mean fluoride concentrations (ug/cm^3) in simulated rainfall (SR) entering each model unit (CS, TSS and TS) and leachate produced from numbered drains of each model unit at each rainfall simulation (standard deviations in parenthesis for means of 2).

Model Unit	Drain No.	Mean Fluoride Concentration (ug/cm^3) of Simulated Rainfall Applied/ /Leachate Produced at Day:								
		15	**19**	**22**	**28**	**32**	**37**	**47**	**52**	**55**
SR		0.146 (0.005)	0.139 (0.004)	0.192 (0.096)	0.101 (0.003)	0.051 (0.023)	0.092 (0.010)	0.122 (0.004)	0.086 (0.001)	0.052 (0.001)
CS	**6, 7, 8**	0.904 (0.022)	0.735 (0.016)	1.319 (0.145)	1.212 (0.038)	0.984 (0.209)	1.186 (0.000)	0.444 (0.008)	0.672 (0.014)	1.000' (0.016)
	9, 5	0.293 (0.004)	0.328 (0.005)	0.612 (0.006)	0.893 (0.027)	1.022 (0.156)	0.587 (0.006)	0.874 (0.054)	0.444 (0.031)	0.480' (0.043)
TSS	**5**	0.575 (0.023)	0.752 (0.023)	1.045 (0.075)	X (-)	0.658 (0.021)	0.677 (0.007)	0.654 (0.040)	0.698 (0.037)	0.672 (0.028)
	6	2.090' (0.150)	X (-)	X (-)	X (-)	X (-)	X (-)	X (-)	X (-)	7.532 (0.419)
	7	1.391 (0.043)	1.536 (0.064)	1.938 (0.065)	2.310' (0.000)	1.330' (0.042)	1.243 (0.081)	1.239 (0.000)	1.018 (0.097)	0.870' (0.048)
	8	0.764 (0.008)	1.456 (0.049)	1.722 (0.112)	1.642 (0.108)	1.566 (0.108)	1.110' (0.032)	1.136 (0.070)	X (-)	1.159 (0.038)
	9	0.478 (0.008)	0.616 (0.000)	0.476 (0.030)	0.817 (0.027)	0.702 (0.015)	0.612 (0.006)	0.603 (0.006)	0.628 (0.063)	0.616 (0.000)
TS	**5**	0.452 (0.012)	0.730' (0.008)	X (-)	0.481 (0.004)	0.334 (0.063)	0.353 (0.031)	0.276 (0.021)	0.444 (0.008)	0.418 (0.028)
	6	0.692 (0.014)	1.064 (0.032)	1.330' (0.042)	1.805 (0.123)	1.818 (0.023)	1.760' (0.059)	1.270' (0.043)	1.456 (0.049)	1.801 (0.129)
	7	0.897 (0.032)	1.041 (0.000)	1.064 (0.032)	1.805 (0.123)	1.171 (0.183)	1.186 (0.000)	0.996 (0.064)	0.677 (0.007)	0.954 (0.048)
	8	0.735 (0.016)	1.064 (0.032)	1.132 (0.000)	1.801 (0.000)	1.086 (0.064)	1.604 (0.054)	0.950' (0.000)	0.912 (0.032)	1.33 (0.129)
	9	0.599 (0.011)	0.583 (0.011)	0.375 (0.000)	1.064 (0.032)	0.342 (0.006)	0.630' (0.006)	0.634 (0.013)	0.481 (0.004)	0.639 (0.019)

X = Blocked Drain/No Collection or Inadequate Volume Collected for Analysis.

Table 3.3.2.4. Quantities of fluoride (conc. x volume) in mg in the total simulated rainfall, SR, and the total leachates collected from model units CS, TSS and TS at each rainfall simulation (see tables 3.3.2.1 and 3.3.2.3) and over all simulations. The quantities for each simulation are expressed both numerically and figuratively with I equivalent to 2.5 mg

Day	Quantity of fluoride (mg) in the total rainfall / leachates from :			
	SR	**CS**	**TSS**	**TS**
15	4.49 II	10.56 IIII	13.9 IIIIII	18.17 IIIIIII
19	4.29 II	10.48 IIII	16.86 IIIIII	18.08 IIIIIII
22	5.93 III	20.08 IIIIIIII	13.6 IIIII	13.37 IIIII
28	4.68 II	34.19 IIIIIIIIIIII	35.5 IIIIIIIIIIIII	49.78 IIIIIIIIIIIIIIIII
32	2.36 I	38.63 IIIIIIIIIIIIII	29.09 IIIIIIIIII	25.76 IIIIIIIII
37	4.26 II	29.6 IIIIIIIIIII	24.82 IIIIIIII	30.27 IIIIIIIIII
47	5.63 III	25.5 IIIIIIIII	25.47 IIIIIIIII	26.37 IIIIIIIII
52	3.96 II	17.42 IIIIIII	24.84 IIIIIIII	23.62 IIIIIIII
55	2.41 I	23.06 IIIIIIII	24.51 IIIIIIII	33.01 IIIIIIIIIII
Total	**38.02**	**209.52**	**208.57**	**238.44**

Table 3.3.2.5. Mean fluoride concentrations (ug/cm^3), over all rainfall simulations (see table 3.3.2.3) in leachates from numbered drains of model units CS, TSS and TS (standard errors in parenthesis for means of up to 18). Values are expressed numerically and figuratively with I equivalent to 0.1 ug/cm .

Model Unit	Drain No.	Mean fluoride concentration. (ug/cm^3)	
CS	6,7,8	0.940 (0.067)	IIIIIIII
	9,5	0.615 (0.061)	IIIII
TSS	5	0.716 (0.035)	IIIIII
	6	4.810 (1.580)	IIIIIIIIIIIIII -> 48
	7	1.431 (0.103)	IIIIIIIIIII
	8	1.319 (0.081)	IIIIIIIIII
	9	0.616 (0.024)	IIIII
TS	5	0.436 (0.034)	IIII
	6	1.444 (0.091)	IIIIIIIIIII
	7	1.088 (0.072)	IIIIIIIII
	8	1.179 (0.079)	IIIIIIIIII
	9	0.594 (0.048)	IIIII

Table 3.3.2.6. Quantities of fluoride (mg) in the total volume (dm^3) of leachates collected from separate drains of each model unit CS, TSS and TS over all rainfall simulations. Quantities collected from drains 5, 6, 7, 8 and 9 of TSS and TS are combined for drains 6,7,8 and 9,5 for comparison with these drains of CS. All values are expressed both numerically and figuratively with I equivalent to 10 mg and 20 dm^3 for fluoride and volume respectively.

Model Unit	Drain No.	Fluoride (mg)	Volume (dm^3)
CS	6,7,8	89.6 IIIIIIII	94.5 IIIII
	9,5	119.9 IIIIIIIIII	188.8 IIIIIIII
TSS	5	11.2 I	16.6 I
	6	1.3	0.4
	7	23.0' II	16.1 I
	8	3.8	3.2
	9	169.2 IIIIIIIIIIIIII	268.6 IIIIIIIIIII
	6,7,8	28.2 III	19.7 I
	9,5	180.3 IIIIIIIIIIIIIII	285.2 IIIIIIIIIIII
TS	5	18.5 II	46.3 II
	6	20.9 II	14.1 I
	7	30.6 III	28.6 I
	8	72.5 IIIIII	59.1 III
	9	96 IIIIIIIII	164.8 IIIIIII
	6,7,8	124 IIIIIIIIIII	101.8 IIIII
	9,5	114.5 IIIIIIIII	211.1 IIIIIIIIII

Table 3.3.2.7. Mean total chromium concentrations (ug/cm^3) in simulated rainfall (SR) entering each model unit (CS, TSS and TS) and leachate produced from numbered drains of each model unit at each rainfall simulation (standard deviations in parenthesis for means of 2).

Model Unit	Drain No.	Mean Total Chromium Concentration (ug/cm^3) of Simulated Rainfall Applied/Leachate Produced at Day:								
		15	**19**	**22**	**28**	**32**	**37**	**47**	**52**	**55**
SR		0.002 (0.002)	0.001 (0.001)	0.006 (0.009)	0.004 (0.002)	0.012 (0.002)	0.016 (0.005)	0.005 (0.007)	0.012 (0.008)	0.011 (0.006)
CS	**6, 7, 8**	0.018 (0.004)	0.012 (0.013)	0.018 (0.009)	0.002 (0.001)	0.014 (0.018)	0.018 (0.011)	0.019 (0.009)	0.019 (0.010)	0.006 (0.008)
	9, 5	0.010' (0.000)	0.022 (0.018)	0.022 (0.011)	0.021 (0.002)	0.020' (0.001)	0.013 (0.014)	0.014 (0.012)	0.006 (0.008)	0.012 (0.011)
TSS	**5**	0.098 (0.021)	0.078 (0.008)	0.068 (0.009)	X (-)	2.908 (0.057)	1.032 (0.043)	0.516 (0.031)	0.367 (0.027)	0.439 (0.006)
	6	0.720' (0.023)	**28.33 (0.311)**	X (-)	X (-)	X (-)	X (-)	X (-)	X (-)	0.781 (0.019)
	7	0.034 (0.013)	0.021 (0.007)	0.021 (0.009)	0.02 (0.006)	0.054 (0.013)	0.048 (0.011)	0.018 (0.003)	0.028 (0.007)	0.021 (0.008)
	8	0.013 (0.008)	0.019 (0.004)	0.024 (0.013)	0.007 (0.007)	0.024 (0.009)	0.061 (0.009)	0.016 (0.017)	0.010' (0.006)	0.028 (0.011)
	9	0.012 (0.009)	0.032 (0.001)	0.122 (0.024)	0.213 (0.006)	0.251 (0.000)	0.160' (0.010)	0.130' (0.011)	0.093 (0.006)	0.108 (0.013)
TS	**5**	0.016 (0.011)	0.008 (0.002)	0.012 (0.002)	0.318 (0.026)	1.110' (0.029)	0.506 (0.011)	0.149 (0.016)	0.060' (0.001)	0.194 (0.013)
	6	0.93 (0.013)	4.389 (0.051)	3.593 (0.044)	2.844 (0.052)	1.002 (0.012)	1.182 (0.064)	0.773 (0.010)	0.432 (0.026)	0.335 (0.001)
	7	0.032 (0.013)	0.301 (0.005)	0.388 (0.023)	0.206 (0.009)	0.086 (0.016)	0.071 (0.004)	0.084 (0.004)	0.034 (0.021)	0.027 (0.014)
	8	0.032 (0.011)	0.088 (0.010)	0.066 (0.018)	0.075 (0.006)	0.040' (0.014)	0.039 (0.001)	0.058 (0.012)	0.032 (0.004)	0.025 (0.017)
	9	0.026 (0.003)	0.075 (0.013)	0.112 (0.005)	0.084 (0.022)	0.134 (0.015)	0.087 (0.003)	0.080' (0.009)	0.058 (0.008)	0.076 (0.018)

X = Blocked Drain/No Collection or Inadequate Volume Collected for Analysis.

Probably due to drain blockage, the unusually high mean total chromium concentration recorded for the the leachates from drain 6 of TSS on day 19 was clearly unrepresentative of concentrations found elsewhere. Hence, this value was excluded from further data manipulations in tables 3.3.2.8, 3.3.2.9 and 3.3.2.10.

Table 3.3.2.8. Quantities of total chromium (conc. x volume) in mg in the total simulated rainfall, SR, and the total leachates collected from model units CS, TSS and TS at each rainfall simulation (see tables 3.3.2.1 and 3.3.2.6) and over all simulations. The quantities for each simulation are expressed both numerically and figuratively with I equivalent to 1 mg.

Day	Quantity of total chromium (mg) in the total rainfall / leachates from :			
	SR	**CS**	**TSS**	**TS**
15	0.08	0.28	0.47	2.39 II
19	0.02	0.44	0.76 I	7.98 IIIIIII
22	0.2	0.49	2.64 III	5.29 IIIII
28	0.16	0.44	7.69 IIIIIII	12.5 IIIIIIIIII
32	0.53 I	0.67 I	13.94 IIIIIIIIIII	10.58 IIIIIIIII
37	0.72 I	0.54 I	7.58 IIIIIII	7.69 IIIIIII
47	0.23	0.55 I	6.4 IIIII	4.36 IIII
52	0.53 I	0.35	4.55 IIIII	2.54 III
55	0.51 I	0.36	5.06 IIIII	3.56 IIII
Total	**2.98**	**4.12**	**49.09**	**56.44**

Total Chromium: - Table 3.3.2.9 shows numerical and figurative expressions of the mean total chromium concentrations in each numbered leachate from each model unit over all rainfall applications.

Total Chromium: - Table 3.3.2.10 displays the concentrations of total chromium in each numbered leachate of CS, TSS and TS (table 3.3.2.7) multiplied by their respective volumes (table 3.3.2.1) and combined for each model unit to give numerical and figurative expressions of the total quantities of total chromium found in the leachates from separate drains of each model unit over all rainfall simulations. The total volume of each separate leachate is also shown.

Chromium (VI): - Table 3.3.2.11 shows the mean chromium (VI) concentrations of freshly collected leachates from each drain of TSS and TS at each rainfall simulation.

Chromium (VI): - Table 3.3.2.12 shows the mean chromium (VI) concentrations of freshly collected leachates from each drain of TSS and TS at each rainfall simulation (table 3.3.2.11), presented as percentages of the mean total chromium concentrations of these leachates (table 3.3.2.7).

Total Chromium / Chromium (VI): - Table 3.3.2.13 displays numerical and figurative expressions of the total quantities of total chromium and chromium (VI) in the total leachates from each model unit TSS and TS at each rainfall simulation. This table also shows these quantities of chromium (VI), presented numerically and figuratively, as percentages of the corresponding quantities of total chromium. Table 3.3.2.14 displays identical expressions of total chromium and chromium (VI) for the leachates from each drain of model units TSS and TS, over all rainfall simulations.

Chromium (VI): - Table 3.3.2.15 shows the mean chromium (VI) concentrations of leachates from each drain of TSS and TS at each rainfall simulation, after ageing periods of

Table 3.3.2.9. Mean total chromium concentrations (ug/cm^3), over all rainfall simulations (see table 3.3.2.7), in leachates from numbered drains of model units CS, TSS and TS (standard errors in parenthesis for means of up to 18). Values are expressed numerically and figuratively with I equivalent to 0.02 ug/cm^3.

Model Unit	Drain No.	Mean total chromium concentration (ug/cm^3).	
CS	6,7,8	0.014 (0.002)	I
	9,5	0.016 (0.002)	I
TSS	5	0.688 (0.230)	IIIIIIIIII -> 34
	6	0.751 (0.020)	IIIIIIIIIII -> 38
	7	0.029 (0.008)	I
	8	0.022 (0.004)	I
	9	0.124 (0.018)	IIIII
TS	5	0.264 (0.082)	IIIIIIIIIII
	6	1.720 (0.341)	IIIIIIIIIIIII -> 86
	7	0.137 (0.030)	IIIIIII
	8	0.050 (0.005)	II
	9	0.081 (0.007)	IIII

Table 3.3.2.10. Quantities of total chromium (mg) in the total volume (dm^3) of leachates collected from separate drains of each model unit CS, TSS and TS over all rainfall simulations. All values are expressed both numerically and figuratively with I equivalent to 5 mg and 20 dm^3 for total chromium and volume respectively.

Model Unit	Drain No.	Total Chromium (mg)		Volume (dm^3)	
CS	6,7,8	1.2		94.5	IIIII
	9,5	2.9	I	188.8	IIIIIIII
TSS	5	12.2	II	16.6	I
	6	0.2		0.4	
	7	0.5		16.1	I
	8	0.1		3.2	
	9	36.2	IIIIIII	268.6	IIIIIIIIIII
TS	5	14.3	III	46.3	II
	6	21.7	IIII	14.1	I
	7	3.6	I	28.6	I
	8	2.9	I	59.1	III
	9	13.9	III	164.8	IIIIIII

Table 3.3.2.11. Mean chromium (VI) concentrations (ug/cm^3) of leachates from numbered drains of model units TSS and TS at each rainfall simulation. Standard deviations in parenthesis are for means of 2.

Model Unit	Drain No.	Mean Chromium (VI) Concentration (ug/cm^3) of Leachate Produced at Day:								
		15	19	22	28	32	37	47	52	55
TSS	5	ND	ND	ND	X	2.642 (0.000)	0.930' (0.000)	0.382 (0.051)	0.202 (0.005)	0.335 (0.008)
	6	0.708 (0.011)	17.675 (0.106)	X	X	X	X	0.744 (0.001)	X	0.749 (0.068)
	7	ND	ND	ND	ND	0.043 (0.001)	0.038 (0.001)	ND	ND	ND
	8	ND	ND	ND	ND	0.017 (0.001)	0.048 (0.008)	ND	X	ND
	9	ND	ND	ND	0.008 (0.001)	0.158 (0.000)	0.075 (0.004)	0.024 (0.004)	ND	ND
TS	5	ND	ND	ND	0.205 (0.010)	1.017 (0.001)	0.506 (0.008)	0.124 (0.001)	ND	0.117 (0.007)
	6	0.812 (0.018)	4.13 (0.001)	3.112 (0.159)	2.702 (0.069)	0.988 (0.000)	1.183 (0.034)	0.760' (0.001)	0.425 (0.003)	0.298 (0.000)
	7	ND	0.213 (0.007)	0.326 (0.008)	0.110' (0.008)	0.081 (0.001)	0.066 (0.007)	0.058 (0.000)	ND	ND
	8	ND	ND	ND	0.042 (0.008)	0.038 (0.001)	0.023 (0.000)	0.028 (0.003)	ND	ND
	9	ND	ND	ND	ND	0.074 (0.003)	0.034 (0.003)	0.008 (0.001)	ND	ND

X = Blocked Drain/No Collection or Inadequate Volume Collected for Analysis.

ND = Not Detected.

Table 3.3.2.12. Mean chromium (VI) concentrations (ug/cm^3) of leachates from numbered drains of model units TSS and TS at each rainfall simulation (table 3.3.2.11), presented as percentages (to the nearest whole number) of the mean total chromium concentrations (ug/cm^3) of these leachates (table 3.3.2.7).

Model Unit	Drain No.	Cr (VI) as a percentage of Cr in leachates Collected at Day:								
		15	19	22	28	32	37	47	52	55
TSS	5					91	90	74	55	76
	6	98	62							96
	7					80	79			
	8					71	79			
	9				4	63	45	18		
TS	5				64	92	100	83		60
	6	87	95	87	95	99	100	98	98	89
	7		71	84	53	94	93	69		
	8				56	95	59	48		
	9					55	39	10		

In TSS, shaded areas represent uncollected leachates due to drain blockage or leachates where chromium (VI) was not detected. In TS, all shaded areas represent leachates where chromium (VI) was not detected (see table 3.3.2.11).

Table 3.3.2.13. Quantities of total chromium and chromium (VI) (mg) in the total leachates from model units TSS and TS at each rainfall simulation and over all simulations, and the percentages of these quantitities of total chromium present as chromium (VI). Values are expressed numerically and figuratively with I equivalent to 1 mg * and 5 % **.

Model Unit	Day of Rainfall Simulation	Quantity of Total Chromium (mg) *	Quantity of Chromium (VI) (mg) *	Percentage of Total Chromium as Chromium (VI) **
TSS	15	0.47	0.08	17.02 III
	19	0.76 I	0.00'	0.00'
	22	2.64 III	0.00'	0
	28	7.69 IIIIIII	0.28	3.64 I
	32	13.94 IIIIIIIIIIII	10.14 IIIIIIII	72.74 IIIIIIIIIIII
	37	7.58 IIIIIII	4.52 IIII	59.63 IIIIIIIII
	47	6.40' IIIII	2.32 II	36.25 IIIIII
	52	4.55 IIII	0.84 I	18.46 III
	55	5.06 IIII	1.32 I	26.09 IIII
	Total	**49.09 mg**	**19.50 mg**	**39.72 %**
TS	15	2.39 II	1.46 I	61.09 IIIIIIIII
	19	7.98 IIIIIII	5.84 IIIII	73.18 IIIIIIIIIII
	22	5.29 IIII	2.70' III	51.04 IIIIIII
	28	12.50' IIIIIIIII	8.64 IIIIIII	69.12 IIIIIIIIII
	32	10.58 IIIIIIIII	8.66 IIIIIIII	81.85 IIIIIIIIIIII
	37	7.69 IIIIIII	6.60' IIIIII	85.82 IIIIIIIIIIII
	47	4.36 IIII	2.54 III	58.26 IIIIIIIII
	52	2.54 III	0.64 I	25.2 IIII
	55	3.56 IIII	1.22 I	34.27 IIIII
	Total	**56.44 mg**	**38.29 mg**	**67.84 %**

Total chromium and chromium (VI) concentrations for TSS exclude those unrepresentative concentrations recorded in the leachates from drain 6 on day 19 (see tables 3.3.2.7 and 3.3.2.11).

Table 3.3.2.14. Quantities of total chromium and chromium (VI) (mg) in individual leachates from model units TSS and TS combined over all rainfall simulations, and the percentages of these quantities of total chromium present as chromium (VI). Values are expressed numerically and figuratively with I equivalent to 2 mg* and 5 %**.

Model Unit	Drain No.	Quantity of Total Chromium (mg) *		Quantity of Chromium (VI) (mg) *		Percentage of Total Chromium as Chromium (VI) **	
TSS	5	12.18	IIIIII	9.82	IIIII	80.64	IIIIIIIIIIII
	6	0.02	(*)	0.02	(**)	97.71	IIIIIIIIIIIIIII
	7	0.47		0.16		34.33	IIIIII
	8	0.08		0.03		40.00'	IIIIII
	9	36.86	IIIIIIIIIIIIIII	9.28	IIIII	25.65	IIIII
TS	5	14.34	IIIIII	11.61	IIIII	80.95	IIIIIIIIIIII
	6	21.73	IIIIIIIII	20.47	IIIIIIII	94.21	IIIIIIIIIIIIII
	7	3.64	II	2.50'	I	68.61	IIIIIIIII
	8	2.87	I	1.07	I	37.35	IIIIII
	9	13.86	IIIIII	2.64	I	19.04	IIII

(*) This value excludes the unrepresentative total Cr concentration on day 19 (see table 3.3.2.7).

(**) This value excludes that Cr (VI) concentration in this leachate on day 47, as no total Cr measurement was made at this time. This value also excludes the unusually high Cr (VI) concentration found in this leachate at day 19 (see table 3.3.2.11).

Table 3.3.2.15. Mean chromium (VI) concentrations (ug/cm^3) of leachates from numbered drains of model units TSS and TS at each rainfall simulation, measured after ageing periods shown (standard deviations in parenthesis for means of 2), and its total quantities extrapolated for the total volume of leachates produced, from each drain of each model unit over all rainfall simulations, from each model unit at each simulation, and from each unit over all simulations.

Model Unit	Drain No.	Mean Chromium (VI) Concentration (ug/cm^3) of Leachate Produced at Day Specified: Measured after Ageing Period (Days) Shown:									Total (mg)
		15 (145)	19 (142)	22 (139)	28 (133)	32 (129)	37 (124)	47 (114)	52 (109)	55 (106)	
TSS	5	ND	ND	ND	X	1.214 (0.023)	0.536 (0.006)	0.302 (0.017)	0.152 (0.005)	0.334 (0.018)	6.14
	6	(X)	(X)	X	X	X	X	(X)	X	0.388 (0.008)	0.05
	7	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.00'
	8	ND	ND	ND	ND	ND	ND	ND	X	ND	0.00'
	9	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.00'
Total (mg)		0.00'	0.00'	0.00'	0.00'	2.00'	1.11	1.19	0.63	1.27	**6.2**
TS	5	ND	ND	ND	0.200' (0.017)	0.501 (0.009)	0.395 (0.003)	ND	ND	ND	7.10'
	6	(X)	1.852 (0.060)	1.427 (0.082)	1.102 (0.011)	0.422 (0.000)	(X)	0.476 (0.007)	0.341 (0.004)	0.306 (0.013)	8.32
	7	ND	0.176 (0.030)	ND	ND	ND	ND	ND	ND	ND	0.51
	8	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.00'
	9	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.00'
Total (mg)		0.00'	2.85	0.83	4.33	2.94	3.25	0.70'	0.52	0.52	**15.94**

X = Blocked Drain/NoCollection or Inadequate Volume Collected for Analysis.

(X) = Chromium (VI) concentration recorded for intial analysis only (table 3.3.2.11).

ND = Not Detected.

up to 145 days after collection. This table also indicates these concentrations presented as quantities (volume x concentration) of chromium (VI) in the separate leachates from each model unit over all rainfall simulations, in the total leachates from each model unit at each rainfall simulation, and as the total quantity of chromium (VI) leached from each model unit.

3.3.2.3. Leachate collection.

3.3.2.3.1. Drain function.

The variation in leachate volumes from identical drains of each model unit when compared between applications of identical volumes of simulated rainfall (table 3.3.2.1) highlighted the inability of the drain design (section 3.2.2.2.) to extract extremely consistent volumes of leachate from individual positions within the soil profiles. Restriction of leachate flow, at various rainfall simulations, from drains 5, 6 and 8 of TSS and drain 5 of TS (table 3.3.2.1) was due to severe blockage by soil and these drains required clearing by manual suction prior to and during each SR application. The lost volumes from drain 6 of TSS were apparently compensated for by greater movement of waters to drain 7 (table 3.3.2.1).

3.3.2.3.2. The total volumes of leachate collected from each model unit.

The total volumes of leachate collected from CS, TSS and TS over all 9 rainfall simulations was approximately 76, 82 and 84 % respectively of the volumes of simulated rainfall applied (based on table 3.3.2.1). This indicated that procedures carried out to maintain the soil of each model unit at field capacity during the rainfall simulations (sections 3.2.3.4. and 3.2.4.3.) were only partially successful. The lower total volume of leachate collected from CS compared to that collected from TSS and TS (table 3.3.2.1.), was found on later examination, to have been caused by a leak which had developed in the wall of this

model unit adjacent to the exit for base drain 5 (figure 3.2.5).

Table 3.3.2.1 shows the expected increase in the total volume of leachates from each model unit as the volume of simulated rainfall was increased by 50 % at the 4th application on day 28 of the experiment. The mean total volume of leachates from model units CS, TSS and TS for the last 6 simulated rainfall applications represented an approximate increase of 52, 55 and 56 % respectively over the mean total volumes for the first 3 applications (based on table 3.3.2.1).

3.3.2.3.3. Partitioning of leachates within the soil profile of each model unit.

As expected for such free draining soils (section 3.2.2.1.), the greatest volume of drainage water of the total collected from each soil profile was collected via the combination of drains 5 and 9 (table 3.3.2.1), situated at the base of each soil profile, and which are respectively the closest and most distant operative drains relative to the base of each pole section (figure 3.2.5). The remainder left each model unit via the combination of drains 6, 7 and 8, situated higher in the profile and respectively the 1st, 2nd and 3rd closest operative drains to the face of each pole section (figure 3.2.5). The larger volumes of drainage water collected from the base drains 5 and 9 represented 66.64 and 67.48 % of the total from model units CS and TS respectively, whereas in TSS this volume represented 93.54 % of the total (based on table 3.3.2.1). Though this latter percentage was obviously enhanced somewhat by the disfunction of drains 6 and 8 in model unit TSS (section 3.3.2.3.1.), sand amendment of the TSS soil profile (section 3.2.2.1.) clearly facilitated more efficient drainage by reducing vertical impedance to water movement. The greater vertical impedance experienced by waters entering the profiles of CS and TS improved lateral water flow in these soil profiles resulting in the relatively greater volumes of soil leachate entering drains 6, 7 and 8 of these model units (table 3.3.2.1).

3.3.2.4. The pH of leachates and simulated rainfall.

Throughout the experiment, the pH of SR and leachates of CS, TSS and TS ranged from weakly acidic to weakly alkaline (table 3.3.2.2). There were generally no significant differences between the mean pH of leachates from CS, TSS and TS, or between these and the pH of SR at each rainfall simulation. Similarly, there were no significant pH differences between leachates from different drains within each model unit or between leachates from identical or similarly situated drains of different model units. Though there were no significant differences between the mean pH of SR (combined for all rainfall simulations) and leachates of CS, TSS and TS (combined for all rainfall simulations and drains) the leachates of model units TSS and TS were of higher mean pH (table 3.3.2.2)

The mean pH of the total leachates from CS at days 22 and 28 was significantly lower than those leachates from this model unit at days 47, 52 and 55, $P < 0.0005$, and the mean pH of total leachates from TSS and TS at days 28 and 32 were significantly lower than those from TSS and TS respectively at all other rainfall simulations, $P < 0.0005$ for both (table 3.3.2.2). These significant differences were clearly associated with the markedly lower pH of the simulated rainfall applied on these days.

3.3.2.5. The fluoride contents of simulated rainfall, SR, and the leachates from each model unit, CS, TSS and TS.

3.3.2.5.1. The quantities of fluoride in SR and the total leachates from CS, TSS and TS.

At each rainfall simulation the quantity of fluoride in the total leachates collected from each model unit was much higher than that in the simulated rainfall applied (table 3.3.2.4). While there were differences between the quantites of fluoride found in the total leachates

from each model unit at each simulation, these differences did not consistently favour any one model unit (table 3.3.2.4). Consequently, the degree and pattern of fluoride contamination in the total leachates from each model unit at each simulation were generally similar, though the total leachates collected from model unit TS over all simulations contained approximately 14 % more fluoride than was found in the total leachates from CS and TSS.

A sharp increase in the quantities of fluoride in the total leachates from each model unit around the rainfall simulation on day 28 (table 3.3.2.4) was clearly due to the increased volumes of simulated rainfall applied from day 28 onwards (section 3.3.2.3.2. and table 3.3.2.1). Though these increased quantities of fluoride decreased quite rapidly thereafter, the quantities of fluoride collected from each model unit at each rainfall simulation after this event remained in excess of those found in these leachates prior to it (table 3.3.2.4).

3.3.2.5.2. Fluoride concentrations in separate leachates from CS, TSS and TS.

The mean fluoride concentration in leachates from drain 6,7,8 of CS, collected over all rainfall simulations, was significantly greater than that from drain 9,5 of this model unit, $P = 0.001$ (table 3.3.2.5). Within model units TSS and TS, the fluoride concentrations in leachates from drain 6 were significantly greater than those in leachates from 7 and 8, which in turn were significantly greater than in leachates from 5 and 9, $P < 0.0005$ for both model units (table 3.3.2.5).

Leachates from drains 6, 7 and 8 of TSS contained significantly greater concentrations of fluoride than that from drain 6,7,8 of CS, with $P < 0.0005$, $P < 0.0005$ and $P = 0.001$ respectively (table 3.3.2.5). However, the fluoride concentrations in leachates from drains 5 and 9 of TSS were not significantly different from that in CS 9,5 (table 3.3.2.5).

Leachates from drains 6 and 8 of TS, contained significantly greater concentrations of fluoride than that from drain 6,7,8 of CS, $P < 0.0005$ and $P = 0.026$ respectively (table 3.3.2.5). However, the fluoride concentration in leachates from drain 9 of TS was not significantly different from that of CS 9,5, and the fluoride concentration in leachates from drain 5 of TS was actually significantly lower than that in CS 9,5, $P = 0.018$ (table 3.3.2.5).

Leachates from drains 5, 6 and 7 of TSS contained significantly greater concentrations of fluoride than the corresponding leachates from TS, $P < 0.0005$, $P < 0.0005$ and $P = 0.010$ respectively, while the fluoride concentrations in leachates from drains 8 and 9 of these model units were not significantly different (table 3.3.2.5).

3.3.2.5.3. The quantities of fluoride in separate leachates from CS, TSS and TS.

The leachates from drain 9,5 of CS collected over all rainfall simulations contained a greater total quantity of fluoride than those leachates from drain 6,7,8 of this model unit (table 3.3.2.6). The significantly greater fluoride concentration of leachates from the latter drain combination (section 3.3.2.5.2.) were more than offset by the much greater volume collected from the former drain combination, such that approximately 57 % of the total quantity of fluoride leached from this model unit was collected from the base drain combination 9,5 (based on table 3.3.2.6).

Within model unit TSS, the fluoride concentrations in leachates from base drains 9 and 5 (section 3.3.2.5.2.) accounted for approximately 81 % and 5 % respectively of the total quantity of fluoride collected from this model unit due to the greater volumes collected from these base drains (based on table 3.3.2.6). In contrast, the leachates from drains 6, 7 and 8 which contained higher fluoride concentrations (section 3.3.2.5.2.) but were generally of much lower volume accounted for approximately 0.5 %, 11 % and 2 % of the total quantity of fluoride collected from this model unit (based on table 3.3.2.6).

In model unit TS, the quantitative imbalance in favour of the base drains found in CS and TSS was reversed, due to the gradual descending order of fluoride quantity in leachates from drains 9, 8, 7, 6 and 5 (based on table 3.3.2.6), as a consequence of the magnitude of significantly greater fluoride concentrations in leachates from the upper profile drains (section 3.3.2.5.2.) more than compensating for the greater combined total volumes of leachate from the base drains (table 3.3.2.6). Hence, in this model unit approximately 52 % of the total quantity of fluoride leached from this model unit was collected from the upper profile drains 6, 7 and 8 (based on table 3.3.2.6).

These characteristic patterns of fluoride distribution ensured that the total quantity of fluoride in leachates collected from the combined upper profile drains of model unit TS was approximately 140 % and 440 % of that in the corresponding leachates of CS and TSS respectively, while the total quantity of fluoride in leachates collected from the combined base drains of model unit TSS was approximately 150 % and 160 % of that in the corresponding leachates of CS and TS respectively (based on table 3.3.2.6).

3.3.2.6. The total chromium contents of simulated rainfall, SR, and the leachates from each model unit, CS, TSS and TS.

3.3.2.6.1. The quantities of total chromium in SR and the total leachates from CS, TSS and TS.

As expected, the lowest quantities of total chromium were generally found in SR at each rainfall simulation; but at 3 simulations the total leachates from model unit CS actually contained less total chromium than SR, whilst the total leachates collected from model units TSS and TS at each rainfall simulation contained similar quantities of total chromium which were greatly in excess of both SR and leachates from CS (table 3.3.2.8). Hence, while the quantity of total chromium found in the total leachates from CS over all rainfall simulations,

at 4.12 mg, represented an increase of approximately 38 % over that found in the total SR applied, this quantity amounted to only approximately 8 % and 7 % of that found in the total leachates from TSS and TS respectively.

An increase in the quantities of total chromium in the total leachates from model units TSS and TS around the rainfall simulation at day 28 (table 3.3.2.8) was evidently due to the increased volumes of simulated rainfall applied from day 28 onwards (section 3.3.2.3.2. and table 3.3.2.1). Though these increases were of short duration, the quantities of total chromium in the leachates collected from TSS at each rainfall simulation afterwards remained in excess of those found prior to day 28, while the quantities of total chromium in the leachates from TS reverted to levels similar to those prior to day 28 (table 3.3.2.8).

3.3.2.6.2. Total chromium concentrations in separate leachates from CS, TSS and TS.

The mean total chromium concentrations in leachates from drains 6,7,8 and 9,5 of CS were not significantly different (table 3.3.2.9). Within TSS, leachates from drains 5 and 6 contained significantly greater mean total chromium concentrations than leachates from drains 7, 8 and 9, $P < 0.0005$, while in TS, leachates from drain 6 contained significantly greater mean total chromium concentrations than leachates from all other drains, $P < 0.0005$ (table 3.3.2.9).

The total chromium concentrations of leachates from drains of TSS or TS, contained significantly greater concentrations than those from similarly situated drains of CS, $P <$ or $= 0.004$, for all comparisons except that of TSS 8 and CS 6,7,8 which were not significantly different (table 3.3.2.9).

Leachates from drain 9 of TSS contained significantly greater total chromium concentrations than the corresponding leachates of TS, $P = 0.031$, while leachates from drains 7 and 8 of TS contained significantly greater total chromium concentrations than the

corresponding leachates of TSS, P = 0.001 and P < 0.0005 respectively (table 3.3.2.9). There were no significant differences between the total chromium concentrations in leachates from drains 5 or 6 of TSS and TS.

3.3.2.6.3. The quantities of total chromium in separate leachates from CS, TSS and TS.

The quantity of total chromium found in the leachates from CS 9,5 was more than double that found in leachates from CS 6,7,8 (table 3.3.2.10). Given the similar mean total chromium concentrations of leachates from each drain combination (section 3.3.2.6.2.), this difference was essentially dictated by the 100 % increase in the total volume of leachates collected from the former drain compared to the latter drain (based on table 3.3.2.10).

In model unit TSS, the quantity of total chromium in leachates from the base drains 5 and 9 made up approximately 25 % and 74 % respectively of the total leached from this model unit (based on table 3.3.2.10) due to their combination of relatively high total chromium concentration (table 3.3.2.9) and greater volume (table 3.3.2.10).

In model unit TS, the more equitable spread of drainage volumes ensured that the quantity of total chromium in leachates from drains 5, 6 and 9 made up approximately 25, 38 and 25 % respectively of the total leached from this model unit (based on table 3.3.2.10).

3.3.2.7. The chromium (VI) content of simulated rainfall, SR, and the leachates from each model unit, CS, TSS and TS, with reference to total chromium.

3.3.2.7.1. Occurrence and distribution of chromium (VI).

Chromium (VI) was not found in SR or leachates from model unit CS. The distribution of chromium (VI) in the drainage waters of model units TSS and TS was distinctly different. In both model units, there appeared to be an initial movement of chromium (VI) from the treated pole section into nearby leachates collected from drains 5 and 6 (table 3.3.2.11). However, whereas in TS there was a progressive movement of chromium (VI) thereafter into leachates from more distant drains, irrespective of their depth within the soil profile, in model unit TSS, the distribution pattern indicated preferential movement of chromium (VI) into leachates from the most distant base drain 9, initially bypassing drains higher in the profile (table 3.3.2.11). In both model units, the occurrence of chromium (VI) in these more distant leachates was brief and by the penultimate rainfall simulation it was not present. However chromium (VI) persisted in the leachates from drains 5 and 6 till the end of the series of simulations (table 3.3.2.11).

In model unit TS, chromium (VI) was generally found in descending order of concentration in leachates from drain 6, 5, 7, 8 and 9, whereas in TSS this order was 6, 5, 9, 7 and 8 (table 3.3.2.11).

In both model units, when chromium (VI) was found in the leachates, it generally made up the bulk of the total chromium present (table 3.3.2.12). This was particularly the case with regard to leachates from drains 5 and 6, situated close to the treated timber, where this species frequently accounted for more than 90 % of the total chromium concentration (table 3.3.2.12). In contrast, for both model units, when chromium (VI) was found in the leachates from drain 9, situated at the greatest distance from the treated timber, it usually accounted for much less than 50 % of the total chromium present (table 3.3.2.12). Despite the loss of

data due to the recurring blockage of drain 6 of TSS, the leachates of both model units displayed identical trends in that the chromium (VI) proportions in each leachate tended to peak around the rainfall simulations on days 32 and 37, and the proportions in each leachate tended to decrease as the distance between the pole section and the drains from which the leachate was collected increased (table 3.3.2.12).

3.3.2.7.2. The quantities of chromium (VI) in the leachates from model units TSS and TS.

With few exceptions the total leachates from model unit TSS at each rainfall simulation contained less chromium (VI) than the corresponding total leachates from TS, and in consequence the total quantity of chromium (VI) leached from TS over all 9 rainfall simulations was almost double that leached from TSS (table 3.3.2.13).

The degree of chromium (VI) contamination of the total leachates from each unit at each rainfall simulation, especially TS, generally coincided closely with that of total chromium in these leachates (table 3.3.2.13). With the exception of those quantities of chromium (VI) in the total leachates from TS on days 52 and 55, the quantity of chromium (VI) in the total leachates from this model unit at each rainfall simulation accounted for the bulk of the total chromium present (table 3.3.2.13). However, in the total leachates from TSS this predominance of chromium (VI) only occurred on days 32 and 37. Hence, the total quantity of chromium (VI) leached from model units TSS and TS over all simulations was 39.72 % and 67.84 % respectively of the similar total quantities of total chromium leached from these model units (table 3.3.2.13).

Approximately 50 % and 48 % of the total quantity of chromium (VI) leached from model unit TSS, was found in the leachates from drains 5 and 9 respectively, which contained the highest quantities of total chromium, and where chromium (VI) accounted for 80.64 % and 25.65 % respectively of the quantity of total chromium present (table

3.3.2.14). In TS, approximately 30 % and 54 % of the total quantity of chromium (VI) leached was found in the leachates from drains 5 and 6 respectively, which likewise contained the highest quantities of chromium, and where chromium (VI) accounted for 80.95 % and 94.21 % respectively of the quantity of total chromium present (table 3.3.2.14).

3.3.2.7.3. The quantities of chromium (VI) in aged leachates from model units TSS and TS.

After an ageing period of up to 145 days, a dramatic decline in the presence of chromium (VI) in leachate samples from both model units was recorded (compare table 3.3.2.15 with tables 3.3.2.11, 3.3.2.13 and 3.3.2.14). The greatest declines were associated with leachates of lower concentration, such as those from drains 7, 8 and 9 of both model units (table 3.3.2.11), resulting in the disappearance of chromium (VI) from leachates 7, 8 and 9 of TSS and from leachates 8 and 9 of TS (table 3.3.2.15). These declines in the total quantities of chromium (VI) in the leachates of each model unit over all rainfall simulations amounted to approximately 70 % and 60 % in TSS and TS respectively (compare tables 3.3.2.13 and 3.3.2.15). In addition, no chromium (VI) was found within the watertable samples of these model units when each was finally drained on day 186/187 of the trial (see section 3.2.4.3.).

3.3.3. Grass sward analyses.

3.3.3.1. Introduction.

Two grass swards were grown consecutively in the soilbed of each model unit (section 3.2.4.4.). The swards downslope of each pole section were sampled according to a sampling plan (figure 3.2.6) and a further sample was taken from the remaining area around each pole section (section 3.2.4.4.). Grass samples from both swards in each model unit were measured for dry weight (section 3.2.5.2.1.) and fluoride and chromium content (section 3.2.5.2.2.). Sampling of the second sward (section 3.2.4.4.2.) included a count of grass leaves emerging through a sampling grid immediately downslope of each pole section (figure 3.2.6).

3.3.3.2. Tables of results.

Table 3.3.3.1 shows the mean dry weights, and fluoride and chromium contents of first and second sward grass samples from the soilbed areas A/B/C, D/E/F, G/H/I and J/K/L (figure 3.2.6) of each model unit CS, TSS and TS.

Table 3.3.3.2 shows the dry weights and fluoride and chromium contents, of first and second sward grass samples from the soilbed within 5 cm of each pole section, outside the sampling area downslope of each pole section (figure 3.2.6), in model units CS, TSS and TS.

Table 3.3.3.3 indicates the mean leaf numbers of second sward grass samples from sample positions 1 - 12, encompassing sample areas A/B/C and D/E/F (figure 3.2.6), of model units CS, TSS and TS.

Table 3.3.3.1. Mean dry weights (g/50 cm^2) and fluoride and chromium contents (ug/g) of first and second sward samples for combined areas ABC, DEF, GHI and JKL of model units CS, TSS and TS (standard deviations in parenthesis for means of 3). All values are also expressed figuratively with I equivalent to 0.02 g/50 cm^2 for dry weight, and 4 ug/g for fluoride and chromium contents.

Parameter	Model	Area	Sward 1		Sward 2	
Mean Dry Wt. of Grass (g/50 cm^2)	CS	A/B/C	0.2546 (0.0126)	IIIIIIIIII	0.2130 (0.0220)	IIIIIII
		D/E/F	0.1985 (0.0198)	IIIIIIII	0.2086 (0.0085)	IIIIIIII
		G/H/I	0.1769 (0.0404)	IIIIIII	0.1403 (0.0260)	IIIIII
		J/K/L	0.1639 (0.0142)	IIIIIII	0.1412 (0.0137)	IIIIII
	TSS	A/B/C	0.1415 (0.0345)	IIIII	0.1302 (0.0181)	IIIII
		D/E/F	0.1469 (0.0240)	IIIIII	0.1714 (0.0094)	IIIIII
		G/H/I	0.1662 (0.0351)	IIIIII	0.1417 (0.0234)	IIIIII
		J/K/L	0.1301 (0.0065)	IIIII	0.1639 (0.0078)	IIIIII
	TS	A/B/C	0.1887 (0.0409)	IIIIIII	0.1402 (0.0467)	IIIIII
		D/E/F	0.2043 (0.0387)	IIIIIII	0.1504 (0.0211)	IIIIIII
		G/H/I	0.2182 (0.0360)	IIIIIIII	0.1405 (0.0098)	IIIIII
		J/K/L	0.1884 (0.0038)	IIIIIII	0.1383 (0.0325)	IIIIII
Mean Fluoride Conc. (ug/g)	CS	A/B/C	04.24 (03.90)	I	06.92 (06.51)	II
		D/E/F	13.73 (01.54)	III	08.35 (07.41)	II
		G/H/I	12.79 (09.44)	III	13.87 (05.91)	III
		J/K/L	03.84 (00.70)	I	09.18 (05.05)	II
	TSS	A/B/C	51.80 (47.20)	IIIIIIIIIII	25.68 (06.28)	IIIII
		D/E/F	14.85 (12.37)	III	19.80 (02.20)	IIII
		G/H/I	18.27 (00.62)	IIIII	24.08 (13.37)	IIIII
		J/K/L	15.42 (07.55)	III	11.37 (03.97)	III
	TS	A/B/C	12.78 (03.45)	III	17.44 (13.50)	IIII
		D/E/F	10.19 (04.77)	III	06.80 (01.12)	II
		G/H/I	11.64 (06.28)	III	22.70 (19.30)	IIIII
		J/K/L	13.93 (07.42)	III	20.20 (-)	IIIII
Mean Chromium Conc. (ug/g)	CS	A/B/C	15.80 (26.30)	IIII	04.83 (04.20)	I
		D/E/F	15.74 (15.50)	IIII	11.50 (20.00)	III
		G/H/I	08.47 (08.61)	II	05.21 (05.30)	I
		J/K/L	11.29 (10.87)	III	07.53 (09.25)	II
	TSS	A/B/C	96.10 (79.50)	IIIIIII 24 ->	10.57 (10.37)	III
		D/E/F	09.53 (11.28)	II	17.75 (04.31)	IIII
		G/H/I	07.53 (05.65)	II	23.10 (20.43)	IIIII
		J/K/L	03.38 (04.79)	I	08.78 (08.79)	II
	TS	A/B/C	21.10 (19.00)	IIIII	20.38 (15.28)	IIIII
		D/E/F	16.70 (29.00)	IIII	35.10 (42.40)	IIIIIII
		G/H/I	19.11 (16.70)	IIIII	16.55 (15.66)	IIII
		J/K/L	18.20 (08.05)	IIIII	04.51 (06.38)	I

Table 3.3.3.2. Dry weights (g) and fluoride and chromium contents (ug/g) of first and second sward grass samples from outside the sampling grid within 5 cm of pole sections in model units CS, TSS and TS. Values are also expressed figuratively with I equivalent to 0.02 g for dry weight and 5 ug/g for fluoride and chromium content.

Model Unit and Sward	Dry Wt. of Sample (g)	Fluoride conc. (ug/g)	Chromium conc. (ug/g)
CS 1	0.1911 IIIIIIII	015.43 III	003.14 I
CS 2	0.2689 IIIIIIIII	015.77 III	015.99 III
TSS 1	0.2451 IIIIIIIII	051.41 IIIIIIII	175.03 IIII 17.5 ->
TSS 2	0.1826 IIIIIII	066.09 IIIIIIIII	294.09 IIII 59 ->
TS 1	0.1906 IIIIIII	031.02 IIIII	058.76 IIIIIIIII
TS 2	0.1826 IIIIIII	023.62 IIII	009.99 II

Table 3.3.3.3. Mean leaf numbers of second sward grass samples for sample positions 1 - 12, and for all sample positions (see figure 3.2.6) of model units CS, TSS and TS (standard deviations and errors* in parenthesis for means of 16 and 192 respectively). Values are also expressed figuratively with I equivalent to 1.

Sample Position	Mean Leaf Number in Model Unit:		
	CS	**TSS**	**TS**
1	6.5 (3.25) IIIII	4.62 (3.12) IIII	11.38 (3.22) IIIIIIII
2	8.69 (3.14) IIIIIII	7.63 (4.08) IIIIII	8.56 (5.14) IIIIIII
3	10.31 (3.28) IIIIIII	4.62 (2.75) IIII	5.75 (3.09) IIIII
4	11.25 (3.49) IIIIIIII	7.19 (3.25) IIIII	3.81 (1.91) III
5	10.81 (4.49) IIIIIIII	6.81 (3.58) IIIII	8.50 (4.62) IIIIII
6	10.56 (4.80) IIIIIIII	8.44 (4.66) IIIIII	7.94 (3.26) IIIIII
7	7.44 (3.05) IIIII	6.00 (2.90) IIIII	8.44 (2.85) IIIIII
8	7.25 (3.62) IIIIII	7.88 (2.73) IIIIII	6.94 (2.67) IIIIII
9	8.12 (2.87) IIIIII	6.56 (2.56) IIIII	7.62 (3.14) IIIIII
10	8.25 (3.04) IIIIII	7.5 (5.10) IIIIII	6.38 (5.24) IIIII
11	9.69 (3.57) IIIIIII	9.19 (3.56) IIIIII	6.94 (2.67) IIIIII
12	12.00 (4.69) IIIIIIIII	9.69 (3.84) IIIIIIII	5.81 (3.58) IIIII
All *	9.24 (3.94) IIIIIII	7.18 (3.81) IIIIII	7.34 (3.93) IIIIII

3.3.3.3. Dry weight yields of grass samples downslope of pole sections.

3.3.3.3.1. First sward.

Within model unit CS, the mean dry weight yield of grass from area A/B/C was significantly greater than those from areas D/E/F, G/H/I and J/K/L, P = 0.008 (table 3.3.3.1). However, within model units TSS and TS there were no significant differences between the yields from these areas (table 3.3.3.1).

The dry weight yield of grass from area A/B/C of model unit CS was significantly greater than the corresponding yields from TSS and TS, P = 0.013 (table 3.3.3.1). However, the grass yields from areas D/E/F and G/H/I of each model unit were not significantly different. The dry weight yield of grass from area J/K/L of model unit TS was significantly greater than the corresponding yield from CS, which in turn was significantly greater than the corresponding yield from TSS, P = 0.004 (table 3.3.3.1.)

3.3.3.3.2. Second sward.

Within model unit CS, the mean dry weight yields of grass from areas A/B/C and D/E/F were significantly greater than those from G/H/I and J/K/L, P = 0.002 (table 3.3.3.1). Within model unit TS there were no significant differences between the grass yields from these areas (table 3.3.3.1). Within TSS, the yield of grass from area D/E/F was significantly greater than those from areas A/B/C and G/H/I (table 3.3.3.1).

The yield of grass from area A/B/C of model unit CS was significantly greater than the corresponding grass yields from TSS and TS, P = 0.035 (table 3.3.3.1). Similarly, the grass yield from area D/E/F of CS was significantly greater than the corresponding yields from TSS and TS, P = 0.007 (table 3.3.3.1). However, the grass yields from areas G/H/I and J/K/L of each model unit were not significantly different.

3.3.3.4. Fluoride and chromium content of grass samples downslope of pole sections.

Though the mean foliar fluoride and chromium contents of the majority of first and second sward grass samples from areas A/B/C, D/E/F and G/H/I of TSS and TS were clearly higher than those of CS, the variability of the individual values making up these means ensured that there were no significant differences for any inter- or intra-model comparisons (table 3.3.3.1).

3.3.3.5. Dry weight yields, fluoride and chromium content of grass samples within 5 cm of pole sections.

3.3.3.5.1. First sward.

The dry weight yield of grass from TSS was greater than those from CS and TS which were very similar (table 3.3.3.2.). The foliar fluoride and chromium concentrations of grass from the TSS soilbed were greatly in excess of those from TS, which in turn were greater than those from CS (table 3.3.3.2).

3.3.3.5.2. Second sward.

The dry weight yield of grass from CS was greater than those from TSS and TS which were identical (table 3.3.3.2.). The foliar fluoride and chromium concentrations of grass from the TSS soilbed were much greater than those from TS, which were generally greater than those from CS (table 3.3.3.2).

3.3.3.6. Sward density of second sward grass samples up to 10 cm downslope of each pole section.

Within sample positions 1 - 6 of CS (figure 3.2.6), the mean leaf number in the grass sample from 1 was significantly lower than those from 3, 4, 5 and 6, P = 0.006 (table 3.3.3.3). Within these sample positions of TSS, the mean leaf numbers in samples from 1 and 3 were significantly lower than that from 6, P = 0.013, and within 1 - 6 of TS, 3, 4 and 6 were significantly lower than 1, and 4 was also significantly lower than 2, 5 and 6, P < 0.0005 (table 3.3.3.3). Within sample positions 7 - 12 of CS (figure 3.2.6), the mean leaf numbers in grass samples from 7, 8, 9 and 10 were significantly lower than that from 12, P = 0.002 (table 3.3.3.3). Within these positions of TSS, the mean leaf number from 7 was significantly lower than that from 12, P = 0.029, and within 7 - 12 of TS, there were no significant differences (table 3.3.3.3). Within each model unit there were no significant differences between the mean leaf numbers in grass samples from positions 1 and 7, 2 and 8, 3 and 9, 4 and 10, 5 and 11, or 6 and 12 (table 3.3.3.3).

Though there were no significant differences between the mean leaf numbers in grass samples from positions 2, 6, 7, 8, 9, 10 or 11 of different model units (table 3.3.3.3), comparison of the mean leaf numbers in grass samples from the remaining positions indicated the following significant differences between CS, TSS and TS; position 1, TS > CS/TSS, position 3, CS > TS/TSS, position 4, CS > TSS > TS, CS > TSS, CS/TSS > TS, with P < 0.0005, < 0.0005, < 0.0005, = 0.037 and < 0.0005 respectively.

The mean leaf number in grass samples, combined for all sample positions 1 - 12 of model unit CS was significantly greater than those of TSS and TS, P < 0.0005 (table 3.3.3.3).

3.3.4. Rye plant measurements.

3.3.4.1. Introduction.

Rye was sown in the soilbed of each model unit (section 3.2.4.5.1.). A count of viable seedlings was carried out in each soilbed (section 3.2.4.5.3.) prior to whole plant removal from the crop canopy for measurements of plant heights, crop densities, leaf production, rooting depths and dry weight yields of shoots and roots (section 3.2.4.5.5.).

3.3.4.2. Tables of results.

Table 3.3.4.1 shows the number of seeds planted in 3 groups of rows (see figure 3.2.7), adjacent to and downslope of the pole section in each model unit, and the number of seedlings produced from these which survived to 3 weeks after seeding.

Table 3.3.4.2 indicates the mean height of the tallest plant part of each plant and the number of plants, within each of six 600 cm^2 sectors of the soilbed downslope of the pole section in each model unit (see figure 3.2.8). The total number of Rye plants within the entire sampling area of each soilbed is also shown.

Table 3.3.4.3 shows the mean number of leaves, mean number of senesced leaves and the mean total length of surviving leaves per plant, within each of six 600 cm^2 sectors of the soilbed downslope of the pole section in each model unit (see figure 3.2.8).

Table 3.3.4.4 indicates the mean dry weight of shoots, the mean length of the longest root and the mean dry weight of the root system per plant, within each of six 600 cm^2 sectors of the soilbed downslope of the pole section in each model unit (see figure 3.2.8).

Table 3.3.4.1 The number of Rye seedlings surviving to 3 weeks after seeding, in 18 rows adjacent to, and downslope of pole sections in model units CS, TSS and TS (see figure 3.2.7). Values are also expressed figuratively with I equivalent to 6.

Model Unit	Seed Rows	Total Seeds Planted		No. Reaching Seedling Stage	
CS	**1 - 4**	32	IIIII	25	IIII
	5 - 11	74	IIIIIIIII	57	IIIIIII
	12 - 18	73	IIIIIIIII	55	IIIIIII
TSS	**1 - 4**	32	IIIII	21	IIII
	5 - 11	74	IIIIIIIII	63	IIIIIII
	12 - 18	73	IIIIIIIII	60	IIIIIII
TS	**1 - 4**	32	IIIII	19	III
	5 - 11	74	IIIIIIIII	51	IIIIIII
	12 - 18	73	IIIIIIIII	57	IIIIIII

Table 3.3.4.2. Mean height of tallest plant part and number of plants within sample sectors of the Rye plant canopy (see figure 3.2.8) in model units CS, TSS and TS (standard deviations and errors+ in parenthesis for means of up to 16 and 65 respectively). Values are also expressed figuratively with I equivalent to 2.

Parameter	Sample Sector	Model Unit					
		CS		**TSS**		**TS**	
Mean Height of Tallest Plant Part (cm)	1	09.93 (5.09)	IIIII	14.96 (9.42)	IIIIII	11.98 (6.49)	IIIIII
	2	09.75 (4.17)	IIIII	12.07 (7.07)	IIIIII	05.53 (5.55)	III
	3	09.93 (5.24)	IIIII	15.49 (5.65)	IIIIIII	07.15 (5.83)	IIII
	4	16.53 (5.04)	IIIIIII	13.28 (6.70)	IIIIII	12.18 (6.75)	IIIIII
	5	13.00 (6.50)	IIIII	13.81 (6.19)	IIIIII	14.35 (5.43)	IIIIIII
	6	11.91 (4.06)	IIIII	16.38 (4.95)	IIIIIII	14.10 (6.31)	IIIIIII
	Mean +	11.77 (0.76)	IIIII	14.26 (0.83)	IIIIII	11.13 (0.91)	IIIII
Number of Plants	1	9	IIII	11	IIIII	9	IIII
	2	10	IIIII	10	IIIII	8	IIII
	3	8	IIII	11	IIIII	8	IIII
	4	9	IIII	9	IIII	10	IIIII
	5	6	III	16	IIIIIII	8	IIII
	6	8	IIII	8	IIII	11	IIIII
	Total	**50**		**65**		**54**	

Table 3.3.4.3. Mean number of leaves, number of senesced leaves and total length of surviving leaves per plant, within sample sectors of the Rye plant canopy (figure 3.2.8) in model units CS, TSS and TS (standard deviations and standard errors+ in parenthesis for means of up to 16 and 65 respectively). Values are also expressed figuratively with I equivalent to 1 (A), 0.5 (B) and 12 cm (C).

Parameter	Sample Sector	Model Unit		
		CS	**TSS**	**TS**
Mean No. of Leaves per Plant (A)	1	6.00 (0.71) IIIII	6.36 (0.81) IIIII	5.78 (0.44) IIIII
	2	6.30 (0.68) IIIII	6.90 (0.57) IIIIII	6.00 (0.54) IIIII
	3	6.00 (0.76) IIIII	6.91 (0.70) IIIIII	6.12 (0.84) IIIII
	4	6.11 (0.60) IIIII	6.22 (0.44) IIIII	5.80 (0.79) IIIII
	5	6.00 (0.63) IIIII	6.62 (0.62) IIIIII	5.88 (0.84) IIIII
	6	6.38 (0.74) IIIII	6.12 (0.84) IIIII	5.64 (0.51) IIIII
	Mean +	6.14 (0.09) IIIII	6.55 (0.09) IIIIII	5.85 (0.09) IIIII
Mean No. of Senesced Leaves per Plant (B)	1	2.56 (0.53) IIIII	2.54 (0.81) IIII	2.22 (0.44) III
	2	2.50 (0.71) IIIII	1.80 (0.63) III	2.38 (0.74) IIII
	3	2.75 (0.71) IIIII	2.18 (0.75) III	2.62 (0.92) IIII
	4	2.56 (0.53) IIIII	2.22 (0.44) III	2.00 (0.47) III
	5	2.00 (0.63) IIII	2.00 (0.82) III	2.12 (0.64) III
	6	2.38 (0.52) IIII	2.12 (0.64) III	1.82 (0.98) III
	Mean +	2.48 (0.09) IIII	2.14 (0.09) III	2.17 (0.10) III
Mean Total Length of Surviving Leaves (cm) per Plant (C)	1	74.22 (15.47) IIIII	091.40 (36.40) IIIIII	87.18 (25.34) IIIII
	2	82.54 (23.68) IIIII	126.58 (29.72) IIIIIIII	78.31 (21.07) IIIII
	3	81.39 (25.23) IIIII	115.90 (36.80) IIIIIII	80.80 (34.00) IIIII
	4	82.70 (31.60) IIIII	095.52 (27.60) IIIIII	94.92 (30.42) IIIIII
	5	91.00 (36.90) IIIIII	126.57 (28.13) IIIIIIII	81.60 (32.70) IIIII
	6	86.80 (31.90) IIIII	100.40 (34.70) IIIIII	87.56 (25.25) IIIII
	Mean +	82.58 (3.75) IIIII	111.28 (4.25) IIIIIII	85.61 (3.75) IIIII

Table 3.3.4.4. Mean dry weight of shoots, length of longest root and dry weight of roots, within sample sectors of the Rye plant canopy (see figure 3.2.8) in model units CS, TSS and TS (standard deviations and standard errors+ in parenthesis for means of up to 16 and 65 respectively). Values are also expressed figuratively with I equivalent to 5 mg (A), 0.5 cm (B) and 0.2 mg (C).

Parameter	Sample Sector	Model Unit CS		Model Unit TSS		Model Unit TS	
Mean Dry Wt. of Shoots (mg) (A)	1	28.4 (10.2)	IIIIII	47.0 (17.3)	IIIIIIII	38.6 (10.3)	IIIIIII
	2	37.4 (14.3)	IIIIII	56.0 (19.9)	IIIIIIIII	31.6 (10.0)	IIIII
	3	33.0 (10.6)	IIIIII	52.0 (18.6)	IIIIIIII	41.4 (21.6)	IIIIII
	4	33.3 (12.2)	IIIIII	39.4 (13.4)	IIIIII	40.2 (11.7)	IIIIII
	5	37.2 (15.6)	IIIIII	63.3 (16.8)	IIIIIIIIII	39.8 (11.7)	IIIIII
	6	34.1 (16.5)	IIIIII	44.1 (20.3)	IIIIIII	37.9 (10.4)	IIIIII
	Mean +	33.8 (1.8)	IIIIII	51.8 (2.4)	IIIIIIII	38.3 (1.8)	IIIIII
Mean Length of Longest Root (cm) (B)	1	3.35 (0.74)	IIIIII	3.92 (1.70)	IIIIII	2.50 (0.62)	IIII
	2	3.90 (0.83)	IIIIII	3.84 (1.03)	IIIIII	3.66 (0.57)	IIIIII
	3	3.66 (0.74)	IIIIII	3.07 (0.60)	IIIII	3.45 (0.07)	IIIIII
	4	3.76 (1.01)	IIIIII	4.09 (2.21)	IIIIII	3.62 (0.62)	IIIIII
	5	3.02 (0.93)	IIIII	4.06 (1.44)	IIIIII	3.57 (0.80)	IIIIII
	6	2.25 (1.25)	III	3.00 (1.40)	IIIII	4.20 (0.71)	IIIIII
	Mean +	3.51 (0.15)	IIIIII	3.75 (0.24)	IIIIII	3.50 (0.16)	IIIIII
Mean Dry Wt. of Roots (mg) (C)	1	1.3 (0.8)	IIIII	2.4 (1.1)	IIIIIIIII	1.5 (0.4)	IIIIII
	2	1.8 (0.7)	IIIIIII	1.8 (0.6)	IIIIIII	1.1 (0.4)	IIIII
	3	1.4 (0.3)	IIIIII	2.5 (1.5)	IIIIIIIII	1.1 (0.2)	IIIII
	4	1.8 (0.8)	IIIIIII	1.9 (0.9)	IIIIIIII	1.8 (0.5)	IIIIIII
	5	1.7 (0.5)	IIIIIII	2.0 (1.4)	IIIIIIII	1.7 (0.2)	IIIIIII
	6	1.2 (0.1)	IIIII	2.2 (1.1)	IIIIIIII	1.6 (1.1)	IIIIII
	Mean +	1.6 (0.1)	IIIIII	2.1 (0.2)	IIIIIIII	1.5 (0.1)	IIIIII

3.3.4.3. Rye seedling viability in each model unit.

A generalised linear model, used to examine the effect of pole section treatment, distance from the pole section and soil type, on seedling viability proportions within the model units (table 3.3.4.1), indicated that none of these factors had a significant effect.

3.3.4.4. Plant growth within the crop canopies of each model unit.

3.3.4.4.1. Crop canopy heights.

Within model unit CS, the crop canopy height in sector 4 was significantly greater than that in sectors 1, 2 and 3, P = 0.043 (table 3.3.4.2), and in TS, the canopy in sectors 5 and 6 was significantly taller than that in sector 2, P = 0.018. However, within model unit TSS the canopy heights of each sector were not significantly different. The canopy in sector 3 of TSS was significantly taller than in this sector of TS, P = 0.010 (table 3.3.4.2), though there were no other significant differences between the canopy heights of corresponding sectors of the 3 model units.

Comparison of the mean plant height measurements, for all sectors of each model unit, showed that the canopy in the sand amended model unit TSS was taller than that in CS and TS, though this difference was only significant between TSS and TS, P = 0.018 (table 3.3.4.2).

3.3.4.4.2. Plant densities.

The crop canopy of model unit TSS contained 30 % and 20 % more individual Rye plants than CS and TS respectively (table 3.3.4.2).

3.3.4.4.3. Number of leaves per plant.

Within each model unit there were no significant differences between the mean numbers of leaves produced per plant in each sector of the canopy (table 3.3.4.3). The mean number of leaves produced per plant in sectors 1, 4 and 6 of different model units were also not significantly different. However, the mean numbers of leaves per plant in sectors 2, 5 and 3 of TSS were significantly greater than those in sectors 2 and 5 of TS and 3 of CS respectively (table 3.3.4.3), $P = 0.012$, 0.032 and 0.029 respectively.

The mean number of leaves per plant, combined for all sectors of the canopy in TSS, was significantly greater than that of CS and TS, $P < 0.0005$ for both.

3.3.4.4.4. Number of senesced leaves per plant.

The mean numbers of senesced leaves per plant in each sector of the canopy within each model unit were not significantly different (table 3.3.4.3). Similarly, the mean numbers of senesced leaves per plant in corresponding sectors of different model units were not significantly different.

The overall mean number of senesced leaves per plant, combined for all sectors of the canopy in CS, was significantly greater than that in TSS only, $P = 0.022$ (table 3.3.4.3).

3.3.4.4.5. Total length of non-senescent leaves per plant.

The mean total lengths of non-senescent leaves per plant in each sector of the canopy within model units CS and TS were not significantly different (table 3.3.4.3). However, in TSS this parameter was significantly greater for the plants in sector 5 than in sector 1, $P = 0.030$. The mean total length of non-senescent leaves per plant in sectors 2, 3 and 5 of TSS were significantly greater than those in sectors 2 and 3 of CS and TS, and 5 of TS

respectively, $P < 0.0005$, $P = 0.042$ and $P = 0.005$ respectively.

The mean total length of non-senescent leaves per plant, over all sectors of the canopy in TSS, was significantly greater than that of CS and TS, $P < 0.0005$ for both (table 3.3.4.3).

3.3.4.4.6. Dry weight of shoots.

Within model units CS and TS there were no significant differences in shoot dry weights, between each sector of the canopy (table 3.3.4.4). In TSS, significantly higher dry weights were recorded for shoots in sector 5 than for those in sector 4, $P = 0.025$. There were no significant differences between the mean dry weights of individual plant shoots in sectors 3, 4 and 6 of different model units, however, the dry weight of plant shoots in sectors 1, 2 and 5 of TSS were significantly greater than those in sectors 1 of CS, and 2 and 5 of both CS and TS, respectively, $P = 0.017$, 0.007 and 0.001 respectively (table 3.3.4.4).

The mean dry weight of plant shoots, over all sectors of the canopy in TSS, was significantly greater than that in CS and TS, $P < 0.0005$.

3.3.4.4.7. Dry weight of roots.

Within each model unit there were no significant differences between the dry weights of plant root systems in each sector of the canopy (table 3.3.4.4). Similarly, there were no significant differences between the dry weights of plant roots in corresponding sectors of different model units (table 3.3.4.4). However, the mean dry weight of plant roots, over all sectors in TSS, was significantly greater than that in CS and TS, $P = 0.005$ (table 3.3.4.4).

3.3.4.4.8. Rooting depths.

Within each model unit there were no significant differences between the rooting depths of plants in each sector of the canopy, adjudged by measurement of the longest root of each plant root system (table 3.3.4.4). Similarly, there were no significant differences between the rooting depths of plants in corresponding sectors of different model units, and none between model units for comparisons of this parameter combined for all sectors of each canopy (table 3.3.4.4).

3.3.5. Chemical analyses of soil profiles.

3.3.5.1. Introduction.

After the model units CS, TS and TSS had been subjected to approximately 6 months of simulated field conditions (table 3.2.2), soil samples were recovered from the soil profile of each model unit (section 3.2.4.6. and figure 3.2.9) and analysed for fluoride and total chromium content (section 3.2.5.4.1.).

3.3.5.2. Tables of results.

Tables 3.3.5.1 and 3.3.5.2 show numerical and figurative expressions of the mean fluoride and mean total chromium concentrations respectively of soil samples recovered from 3 depths within the soilbeds of model units CS, TSS and TS (figure 3.2.9). The tables include background measurements of mean fluoride and total chromium concentration, carried out for the soil of each model unit before pole section insertion.

3.3.5.3. Soil fluoride.

3.3.5.3.1. Background fluoride concentrations prior to the experiment.

The background or initial fluoride concentration of the soil from model unit TSS was significantly lower than that of CS and TS, $P = 0.026$ and 0.021 respectively (table 3.3.5.1). The background fluoride concentrations of TS and CS were not significantly different.

Table 3.3.5.1. Mean fluoride concentrations (ug/g) of soil samples recovered from various sample positions and depths within the soil profiles of model units CS, TSS and TS (standard deviations in parenthesis for means of 2 and 3*). Values are also expressed figuratively with I equivalent to 60 ug/g.

Sample Depth (cm)	Sample Position	Mean fluoride concentration (ug/g) in soil from model unit:		
		CS	**TSS**	**TS**
0 - 15	**1 (1,3)**	329.50 (020.30) IIIII	385.00 (079.20) IIIII	956.21 (003.38) IIIIIIIIIIII
	2	Not Sampled	161.36 (003.46) III	260.80 (086.80) IIII
	3	Bulked with 1	568.20 (061.50) IIIIIII	856.50 (098.90) IIIIIIIII
	4	Not Sampled	184.20 (014.70) III	335.40 (029.20) IIIII
	5	296.10 (039.20) IIIII	170.87 (002.01) III	350.10 (081.50) IIIII
	6	345.71 (005.66) IIIII	181.86 (001.02) III	370.49 (006.10) IIIII
15 - 30	**1 (1,3)**	301.20 (023.60) IIIII	373.80 (044.30) IIIII	810.70 (084.40) IIIIIIIII
	2	Not Sampled	208.60 (015.00) III	242.70 (133.00) IIII
	3	Bulked with 1	490.00 (035.60) IIIIIII	973.80 (047.50) IIIIIIIIIII
	4	Not Sampled	309.90 (019.40) IIIII	383.30 (000.10) IIIII
	5	377.80 (055.90) IIIII	148.10 (034.90) II	307.90 (132.50) IIIII
	6	311.59 (006.21) IIIII	170.62 (008.06) III	326.20 (035.60) IIIII
30 - BASE	**1,3**	390.40 (038.40) IIIIII	682.45 (007.78) IIIIIIII	631.30 (024.20) IIIIIIII
	4	Not Sampled	322.90 (122.60) IIIII	329.04 (007.90) IIIII
	5	325.80 (128.00) IIIII	299.00 (050.90) IIIII	323.70 (049.80) IIIII
	6	295.44 (009.57) IIIII	327.74 (010.47) IIIII	366.03 (004.88) IIIII
Background *		317.90 (072.80) IIIII	151.40 (041.20) III	280.40 (043.90) IIII

Table 3.3.5.2. Mean chromium concentrations (ug/g) of soil samples recovered from various sample positions and depths within the soil profiles of model units CS, TSS and TS (standard deviations in parenthesis for means of 2 and 3*). Values are also expressed figuratively with I equivalent to 20 ug/g.

Sample Depth (cm)	Sample Position	Mean chromium concentration (ug/g) in soil from model unit:					
		CS		TSS		TS	
0 - 15	1 (1,3)	70.39 (03.97)	IIII	062.09 (12.23)	III	187.79 (09.10)	IIIIIIII
	2	Not Sampled		038.22 (01.14)	II	109.68 (03.61)	IIIII
	3	Bulked with 1		082.60 (32.80)	IIII	217.61 (01.00)	IIIIIIIII
	4	Not Sampled		033.05 (00.93)	II	122.30 (35.30)	IIIII
	5	77.48 (08.28)	IIII	030.14 (06.70)	II	073.81 (04.14)	IIII
	6	77.60 (21.90)	IIII	049.99 (02.49)	II	080.80 (19.10)	IIII
15 - 30	1 (1,3)	79.20 (27.00)	IIII	049.69 (02.45)	II	182.02 (10.71)	IIIIIII
	2	Not Sampled		040.14 (04.70)	II	088.00 (19.90)	IIII
	3	Bulked with 1		050.98 (01.32)	III	155.61 (04.40)	IIIIIII
	4	Not Sampled		045.08 (06.62)	II	073.60 (17.90)	IIII
	5	72.68 (13.88)	IIII	039.38 (02.52)	II	085.53 (09.02)	IIII
	6	59.03 (05.30)	III	045.68 (01.29)	II	101.49 (12.73)	IIIII
30 - BASE	1,3	79.08 (09.55)	IIII	170.80 (06.51)	IIIIII	148.99 (02.38)	IIIIII
	4	Not Sampled		117.44 (00.38)	IIIII	106.52 (08.50)	IIIII
	5	62.72 (00.38)	III	087.40 (15.40)	IIII	094.19 (08.34)	IIIII
	6	58.52 (01.30)	III	080.90 (00.11)	IIII	121.10 (15.40)	IIIII
Background *		50.63 (12.95)	III	034.06 (04.24)	II	058.19 (12.35)	III

3.3.5.3.2. Fluoride concentrations in each soilbed after the experiment.

Within model unit CS there were no significant differences between the fluoride concentrations of soil samples from identical depths (table 3.3.5.1). The fluoride concentration in sample 6 from 0 - 15 cm depth of CS was significantly greater, $P = 0.014$, than those of the corresponding soil samples from the 2 lower profile depths of this model unit (table 3.3.5.1). There were no further significant differences for comparisons of the fluoride concentrations in corresponding samples from different depths within this model unit.

At sample depth 0 - 15 cm in model unit TSS, the fluoride concentration of soil sample 3 > 1 > 2, 4, 5 and 6, $P < 0.0005$ (table 3.3.5.1). Similarly, at sample depth 15 - 30 cm within this model unit, soil sample 3 > 1, 4 > 2, 5 and 6, $P < 0.0005$, and at depth 30 cm - base, the fluoride concentration of soil sample 1/3 > 4, 5 and 6, $P = 0.012$ (table 3.3.5.1). The fluoride concentrations of soil samples 1/3, 5 and 6 from the lower sampling depth in this soilbed were significantly greater than those of samples 1, 5 and 6 respectively from the shallower sampling depths, $P <$ or $= 0.045$ (table 3.3.5.1). The fluoride concentrations of soil samples 2 and 4 from depth 15 - 30 cm were significantly greater, $P <$ or $= 0.049$, than those of the corresponding shallowest soil samples (table 3.3.5.1).

Within model unit TS, the pattern of significant differences was virtually identical to TSS (table 3.3.5.1). At sample depths 0 - 15 cm and 15 - 30 cm the fluoride concentration of samples 1 and 3 were greater than those of all other samples, $P = 0.001$ for both depths, and at the lowest sample depth, sample 1/3 was significantly greater than 4, 5 and 6, $P = 0.001$. However, in model unit TS, only the fluoride concentrations of soil sample 1, 1/3 at the greatest depth, displayed significant differences between sampling depths, such that 0 - 15 cm > 15 - 30 cm > 30 cm - base, $P = 0.018$ (table 3.3.5.1).

Comparison of identical soil samples, ie. same sample position and sample depth, between CS and TSS, indicated that the fluoride concentrations of samples 5 and 6 from CS at the 2 shallower depths were greater than those of the corresponding samples from TSS, $P < \text{or} = 0.046$ (table 3.3.5.1). However, there were no significant differences between TSS and CS for similar comparisons of fluoride concentrations in soil samples 5 and 6 at the lowest depth or in soil sample 1 and 1/3 at the shallower depths, and the fluoride concentration of soil sample 1/3 from TSS at depth 30 cm - base was significantly greater than the corresponding sample from CS, $P = 0.009$ (table 3.3.5.1).

The fluoride concentrations of soil samples 5 and 6 from the 2 shallower depths and sample 5 from the greatest depth of model unit TS were not significantly different from those in corresponding samples from CS (table 3.3.5.1). However, the fluoride concentration of sample 6 from the lowest depth of TS was significantly greater than that of CS, $P = 0.011$, and soil sample 1 from the shallower depths and 1/3 from the greatest depth of TS contained significantly greater fluoride concentrations than the corresponding soil samples of CS, $P < \text{or} = 0.017$.

Similar comparisons between model units TSS and TS showed that, except for samples 2 and 5 at the shallower depths and samples 1/3, 4 and 5 at 30 cm - base, which were not significantly different, the fluoride concentrations of soil samples from TS were all significantly greater than those of TSS, $P < \text{or} = 0.043$ (table 3.3.5.1).

3.3.5.4. Soil chromium.

3.3.5.4.1. Background chromium concentrations prior to the experiment.

The background or initial chromium concentration of the soil from TSS was significantly lower than that of TS, P = 0.033, but not of CS (table 3.3.5.2). The background chromium concentrations of TS and CS were not significantly different.

3.3.5.4.2. Chromium concentrations in each soilbed after the experiment.

Within model unit CS there were no significant differences between the total chromium concentrations of soil samples from identical depths or between identically numbered samples from different depths (table 3.3.5.2).

Similarly, within TSS, at sample depths 0 - 15 cm and 15 - 30 cm there were no significant differences between the chromium concentrations of soil samples, though samples 1 and 3 from depth 0 - 15 cm contained noticeably more chromium than other samples at this depth (table 3.3.5.2). At the 30 cm - base depth, the fluoride concentration of soil sample 1/3 > 4 > 5 and 6, P = 0.001. Within this soilbed, the chromium concentrations of soil samples 1/3, 4, 5 and 6 from the lower sampling depth were significantly greater than the chromium concentrations of samples 1, 4, 5 and 6 from the shallower sampling depths, P < or = 0.019 (table 3.3.5.2).

Within model unit TS, at sample depth 0 - 15 cm, the chromium concentrations of samples 1 and 3 were greater than those of all other samples, the chromium concentration of sample 4 > 5 and 6, and 2 > 5, P = 0.001 (table 3.3.5.2). At sample depth 15 - 30 cm the chromium concentration of samples 1 and 3 were greater than those of all other samples, P = 0.001, and at the lowest sample depth, sample 1/3 was significantly greater than 4 and 5, P = 0.019 (table 3.3.5.2). In this model unit, the chromium concentration of soil sample 1 at

each of the shallower sampling depths was significantly higher than that of soil sample 1/3 at depth 30 cm - base, P = 0.033 (table 3.3.5.2), and the chromium concentration of soil sample 3 at the shallowest sampling depth was greater than that for this sample from depth 15 - 30 cm, P = 0.003.

Comparison of identical soil samples between CS and TSS indicated that, though there were no significant differences for most comparisons, the chromium concentration of sample 5 from CS at the shallowest depth was greater, P = 0.024, than that of the corresponding sample from TSS (table 3.3.5.2), and the chromium concentrations of soil samples 1/3 and 6 from TSS at depth 30 cm - base were significantly greater, P < or = 0.008, than corresponding samples from CS.

The chromium concentrations of samples 1/3, 5 and 6 from the lowest sampling depth of TS and sample 6 from the depth 15 - 30 cm of this model unit were significantly greater, P < or = 0.038, than corresponding samples from CS (table 3.3.5.2). Soil sample 1 from the 2 shallower depths of TS also contained significantly greater chromium concentrations than the corresponding soil samples of CS, P < or = 0.038.

Similar comparisons between model units TSS and TS showed that, except for soil samples 4 and 6 at sampling depth 0 - 15 cm, sample 4 at depth 15 - 30 cm and samples 1/3, 4, 5 and 6 at 30 cm - base, which were not significantly different, the chromium concentrations of soil samples from TS were all significantly greater than those of TSS, P < or = 0.028 (table 3.3.5.2).

3.3.6. Dehydrogenase activity of surface soil.

3.3.6.1. Introduction.

Soil samples were recovered from the topsoil of model units CS, TSS and TS (section 3.2.4.6. and figure 3.2.9). Dehydrogenase activity was measured in representative supplemented and unsupplemented sub-samples of each topsoil sample after incubation for 0, 1, 3 and 4 weeks (section 3.2.5.4.2.). Supplementation was effected using sterilised Rye meal which showed no dehydrogenase activity (section 3.2.5.4.2.).

3.3.6.2. Tables of results.

Tables 3.3.6.1 and 3.3.6.2 show numerical and figurative expressions of the mean dehydrogenase activities of unsupplemented and supplemented topsoil samples respectively (figure 3.2.9), from model units CS, TSS and TS, after 0, 1, 3 and 4 weeks incubation.

Table 3.3.6.3 shows numerical and figurative expressions of the mean dehydrogenase activity of unsupplemented and supplemented topsoil samples, combined for each sample position (figure 3.2.9) of each model unit CS, TSS and TS, over all weeks of incubation.

3.3.6.3. Dehydrogenase activity of unsupplemented soil.

The dehydrogenase activities of unsupplemented soil samples from each model unit remained generally stable over the 4 weeks of measurement and the activities of soil samples from TSS were invariably lower than those in samples from CS and TS (table 3.3.6.1.).

Table 3.3.6.1. Mean dehydrogenase activity (umol $TTFg^{-1}min^{-1} \times 10^{-5}$) of soil samples recovered from model units CS, TSS and TS, and incubated for up to 4 weeks without supplementation (standard deviations in parenthesis for means of 3). Values are also expressed figuratively with I equivalent to 0.4 umol $TTFg^{-1}min^{-1} \times 10^{-5}$.

Sample Time (weeks)	Sample Position	Mean Soil Dehydrogenase Activity (umol $TTFg^{-1}min^{-1} \times 10^{-5}$)					
		CS		TSS		TS	
0	1 (1,3)	4.496 (1.073)	IIIIIIIII	2.495 (0.147)	IIIII	2.953 (0.171)	IIIIII
	2	Not Sampled		2.717 (0.071)	IIIIII	3.485 (0.164)	IIIIIII
	3	Bulked with 1		2.407 (0.118)	IIIII	2.595 (0.130)	IIIII
	4	Not Sampled		2.770 (0.178)	IIIIII	3.475 (0.459)	IIIIIII
	5	3.993 (0.139)	IIIIIIII	2.699 (0.042)	IIIIII	3.287 (0.048)	IIIIII
	6	3.820 (0.722)	IIIIIIII	2.620 (0.075)	IIIIII	3.486 (0.255)	IIIIIII
1	1 (1,3)	4.047 (0.953)	IIIIIIII	2.487 (0.213)	IIIII	3.940 (0.477)	IIIIIIII
	2	Not Sampled		2.559 (0.103)	IIIII	4.372 (0.332)	IIIIIIIII
	3	Bulked with 1		2.147 (0.217)	IIII	2.170 (0.268)	IIII
	4	Not Sampled		2.854 (0.482)	IIIIII	3.825 (0.428)	IIIIIIII
	5	3.217 (0.294)	IIIIIII	2.402 (0.041)	IIIII	4.179 (0.135)	IIIIIIII
	6	2.755 (0.078)	IIIIII	2.254 (0.081)	IIIII	4.106 (0.839)	IIIIIIII
3	1 (1,3)	4.452 (0.393)	IIIIIIII	2.436 (0.105)	IIIII	3.333 (0.098)	IIIIII
	2	Not Sampled		2.706 (0.328)	IIIIII	4.120 (0.320)	IIIIIIII
	3	Bulked with 1		2.378 (0.068)	IIIII	2.905 (0.395)	IIIIII
	4	Not Sampled		2.940 (0.115)	IIIIII	6.188 (0.691)	IIIIIIIIIII
	5	4.786 (0.529)	IIIIIIIII	2.604 (0.130)	IIIIII	5.058 (0.627)	IIIIIIIII
	6	4.031 (0.179)	IIIIIIII	2.516 (0.123)	IIIII	5.475 (0.390)	IIIIIIIIII
4	1 (1,3)	4.709 (0.036)	IIIIIIIII	2.042 (0.136)	IIII	3.038 (0.171)	IIIIII
	2	Not Sampled		2.536 (0.197)	IIIII	4.063 (0.145)	IIIIIII
	3	Bulked with 1		2.847 (0.198)	IIIIII	2.956 (0.142)	IIIIII
	4	Not Sampled		2.716 (0.162)	IIIIII	4.084 (0.636)	IIIIIII
	5	LOST		2.745 (0.064)	IIIIII	4.417 (0.524)	IIIIIIII
	6	3.789 (0.573)	IIIIII	2.666 (0.085)	IIIII	3.616 (0.432)	IIIIIII

Table 3.3.6.2. Mean dehydrogenase activity (umol TTF g^{-1} min^{-1} x 10^{-5}) of soil samples recovered from model units CS, TSS and TS, supplemented with Rye meal and incubated for up to 4 weeks (standard deviations in parenthesis for means of 3). Values are also expressed figuratively with I equivalent to 5 umol TTF g^{-1} min^{-1} x 10^{-5}.

Sample Time (weeks)	Sample Position	Mean Soil Dehydrogenase Activity (umol TTF g^{-1} min^{-1} x 10^{-5})					
		CS		TSS		TS	
0	1 (1,3)	16.134 (1.464)	III	08.879 (0.140)	II	14.600 (1.110)	III
	2	Not Sampled		14.310 (0.454)	III	19.224 (1.550)	IIII
	3	Bulked with 1		04.733 (0.350)	I	06.804 (1.229)	I
	4	Not Sampled		11.356 (1.691)	II	17.777 (1.502)	III
	5	19.150 (1.860)	IIII	12.940 (2.260)	III	21.623 (0.935)	IIII
	6	13.110 (1.306)	III	09.221 (0.698)	II	16.580 (2.310)	III
1	1 (1,3)	55.220 (8.900)	IIIIIIIII	33.022 (1.031)	IIIIII	51.260 (2.820)	IIIIIIII
	2	Not Sampled		39.650 (2.030)	IIIIIII	53.530 (7.790)	IIIIIIIII
	3	Bulked with 1		22.830 (4.360)	IIII	32.910 (12.370)	IIIIII
	4	Not Sampled		46.470 (4.270)	IIIIIII	48.400 (8.920)	IIIIIII
	5	58.160 (2.260)	IIIIIIIII	40.380 (4.630)	IIIIII	42.789 (0.843)	IIIIII
	6	44.430 (0.266)	IIIIIII	41.510 (1.780)	IIIIII	53.880 (6.440)	IIIIIIII
3	1 (1,3)	51.730 (2.050)	IIIIIII	15.521 (0.420)	III	18.316 (0.591)	IIII
	2	Not Sampled		26.430 (4.130)	IIII	47.620 (4.400)	IIIIIIII
	3	Bulked with 1		13.558 (1.353)	III	20.229 (1.470)	IIII
	4	Not Sampled		25.740 (1.701)	IIII	43.940 (2.620)	IIIIIII
	5	37.450 (6.090)	IIIIII	17.801 (1.282)	IIII	52.080 (4.450)	IIIIIII
	6	31.810 (2.530)	IIIII	22.270 (1.890)	IIII	33.114 (1.262)	IIIIII
4	1 (1,3)	26.381 (1.343)	IIII	09.518 (0.132)	II	22.930 (2.660)	IIII
	2	Not Sampled		13.905 (1.019)	III	11.716 (1.528)	II
	3	Bulked with 1		10.552 (0.423)	II	10.386 (1.671)	II
	4	Not Sampled		14.677 (0.524)	III	13.729 (1.657)	III
	5	19.220 (2.860)	IIII	17.597 (0.610)	IIII	23.990 (2.630)	IIII
	6	LOST		15.439 (0.234)	III	22.840 (2.550)	IIII

Table 3.3.6.3. Mean dehydrogenase activities (umol TTFg $^{-1}$ min $^{-1}$ x 10 $^{-5}$) of supplemented and unsupplemented soil samples from model units CS, TSS and TS, over the entire incubation period, tables 3.3.6.1. and 3.3.6.2. (standard deviations in parenthesis for means of 12). Values are also expressed figuratively with I equivalent to 0.4 and 5 umol TTFg $^{-1}$ min $^{-1}$ x 10 $^{-5}$ for unsupplemented and supplemented samples respectively.

Model Unit	Sample Position	Mean Soil Dehydrogenase Activity (umol TTFg $^{-1}$ min $^{-1}$ x 10 $^{-5}$)			
		Unsupplemented soil		**Supplemented soil**	
CS	1,3	4.426 (0.682)	IIIIIIIII	37.37 (17.75)	IIIIII
		–		–	
		–		–	
		–		–	
	5	3.999 (0.747)	IIIIIIII	33.49 (17.08)	IIIIII
	6	3.599 (0.655)	IIIIIII	29.78 (13.72)	IIIII
TSS	1	2.365 (0.237)	IIIII	16.74 (10.20)	III
	2	2.629 (0.192)	IIIIII	23.57 (11.21)	IIII
	3	2.445 (0.298)	IIIII	12.92 (07.11)	III
	4	2.820 (0.251)	IIIIII	24.56 (14.49)	IIII
	5	2.613 (0.152)	IIIIII	22.18 (11.39)	IIII
	6	2.514 (0.185)	IIIII	22.11 (12.71)	IIII
TS	1	3.316 (0.466)	IIIIIII	26.78 (15.18)	IIIII
	2	4.010 (0.403)	IIIIIIII	33.02 (19.08)	IIIIII
	3	2.657 (0.394)	IIIIII	17.58 (11.86)	IIII
	4	4.393 (1.206)	IIIIIIIII	30.96 (16.55)	IIIIII
	5	4.235 (0.725)	IIIIIIIII	35.12 (13.53)	IIIIII
	6	4.171 (0.937)	IIIIIIII	31.60 (15.11)	IIIIII

Within model unit CS there were no significant differences between the dehydrogenase activities of soil samples at weeks 0, 1 and 3 (table 3.3.6.1). However, at week 4 the mean soil dehydrogenase activity of soil from sample position 1 of this model unit was significantly greater than that of sample 6, $P = 0.050$ (table 3.3.6.1).

Within TSS soil samples, at week 0, the dehydrogenase activity of soil sample 3 was significantly lower than those of samples 2 and 4, $P = 0.016$, and at week 1, the activities of samples 1, 5 and 6 were significantly lower than that of sample 4, $P = 0.044$ (table 3.3.6.1). At week 3, the activities of samples 1 and 3 from TSS were significantly lower than that of sample 4, $P = 0.015$, and at week 4, the activity of soil sample 1 of TSS was significantly lower than all other samples from this model unit, $P < 0.0005$ (table 3.3.6.1.).

Within TS soil samples, at week 0, the dehydrogenase activity of soil from sample position 3 of TS was significantly lower than those of samples 2, 4, 5 and 6, $P = 0.003$, and at week 1, the activity of soil sample 3 was significantly lower than all other samples, $P = 0.001$ (table 3.3.6.1). At week 3, the activities of soil samples 1 and 3 of TS were significantly lower than those of samples 4, 5 and 6, and samples 2, 4, 5 and 6 respectively, $P = 0.015$, and at week 4, the activities of soil samples 1 and 3 were significantly lower than all other samples from this model unit, $P < 0.003$ (table 3.3.6.1).

The mean dehydrogenase activity of soil from sample position 1 of CS, over all weeks of measurement, was significantly greater than that of soil sample 6 from this model unit, $P = 0.023$ (table 3.3.6.3). Within TSS, the mean dehydrogenase activity of soil sample 1, combined for weeks 0, 1, 3 and 4, was significantly lower than those of soil samples 2, 4 and 5, the activity of sample 3 was significantly lower than that of 4, and the activity of sample 6 was lower than sample 4, $P < 0.0005$ (table 3.3.6.3). Within model unit TS, the mean dehydrogenase activity of soil sample 3, combined for weeks 0, 1, 3 and 4, was significantly lower than those of soil samples 2, 4, 5 and 6, and the activity of sample 1 was significantly lower than those of 4, 5 and 6, $P < 0.0005$ (table 3.3.6.3).

The mean dehydrogenase activity of soil sample 1 (bulked with 3) of CS, over all weeks was significantly greater than those of the corresponding samples from TSS and TS, $P < 0.0005$ (table 3.3.6.3). In addition, the mean dehydrogenase activities of soil samples 5 and 6 from CS, over all weeks, were significantly greater than the corresponding samples from TSS, $P < 0.0005$ for both (table 3.3.6.3). Similarly, the mean dehydrogenase activities of soil samples 1, 2, 4, 5 and 6 from TS, over all weeks, were significantly greater than those of the corresponding soil samples from TSS, $P < 0.0005$ for each, while the dehydrogenase activities of soil sample 3 of TSS and TS were not significantly different (table 3.3.6.3).

3.3.6.4. Dehydrogenase activity of supplemented soil.

Supplementation with Rye meal, as expected, greatly increased the mean dehydrogenase activities of all soil samples irrespective of the time of measurement (tables 3.3.6.1, 3.3.6.2, and 3.3.6.3). The dehydrogenase activities of supplemented soil samples from each model unit displayed an initial increase from weeks 0 to 1 decreasing then at weeks 3 and 4 (table 3.3.6.2.). The activities of supplemented soil samples from TSS were generally lower than those in samples from CS and TS (table 3.3.6.2.), as with the unsupplemented samples (table 3.3.6.1).

At week 0, the mean dehydrogenase activity of sample 5 of CS was significantly greater than sample 6 of this model unit, $P = 0.009$, and at week 1 the activities of samples 1 and 5 were significantly greater than that of sample 6, $P = 0.043$ (table 3.3.6.2). At week 3 the activity of sample 1 of CS was significantly greater than those of samples 5 and 6, $P = 0.002$, and at week 4 sample 1 displayed significantly greater dehydrogenase activity than sample 5, $P = 0.017$ (table 3.3.6.2).

Within soil samples from TSS all significant differences occurred with $P < 0.0005$. At weeks 0, 1 and 4 the dehydrogenase activity of soil sample 3 of TSS was, with the

exception of sample 1 at week 4, significantly lower than all other samples from this model unit, and at week 3 the dehydrogenase activity of this sample was significantly lower than those of samples 2, 4 and 6 (table 3.3.6.2). At week 0, the dehydrogenase activity of soil sample 1 was significantly lower than samples 2 and 5; at week 1 the activity of soil sample 1 was significantly lower than those of samples 2, 4, 5 and 6; at week 3 the activity of sample 1 was significantly lower than samples 2, 4 and 6; and at week 4, the activity of this sample was significantly lower than all other samples except sample 3 (table 3.3.6.2). The few remaining significant differences between the dehydrogenase activities of soil samples from TSS at weeks 0, 1, 3 and 4 indicated no particular trends.

At weeks 0 and 3 the dehydrogenase activity of soil from sample position 3 of TS was, with the exception of sample 1 at week 3, significantly lower than those of all other samples from this model unit, $P < 0.0005$ at each week (table 3.3.6.2). At week 1, the activity of soil sample 3 was significantly lower than samples 1, 2 and 6, $P = 0.038$, and at week 4 the activity of this soil sample was significantly lower than those of samples 1, 5 and 6, $P < 0.0005$ (table 3.3.6.2). At week 0, the activity of soil sample 1 of TS was significantly lower than those of samples 2 and 5, $P < 0.0005$, and at week 3, the dehydrogenase activity of sample 1 was significantly lower than those of all other samples from this model unit (with the exception of sample 3) $P < 0.0005$. However, at week 4 the activity of sample 1 was significantly greater than those of samples 2, 3 and 4, $P < 0.0005$ (table 3.3.6.2). Other significant differences between dehydrogenase activities of soil samples from TS at weeks 0, 1, 3 and 4 indicated no particular trends.

Within each model unit, the mean dehydrogenase activities of supplemented soil samples, over all weeks, were not significantly different (table 3.3.6.3), due to the large standard deviations introduced by the changing activities in these samples from week to week (table 3.3.6.2). Similarly, apart from the mean dehydrogenase activity of soil sample 1 from TSS, combined for weeks 0, 1, 3 and 4 (table 3.3.6.3), which was significantly lower than that of the corresponding sample from CS, $P = 0.006$, there were no significant

differences between corresponding supplemented soil samples from different model units.

3.3.7. Analyses of the remedially treated pole sections.

3.3.7.1. Introduction.

After exposure to approximately 6 months of simulated field conditions, the 3 pole sections were removed from the soilbeds (section 3.2.4.7). Wood sawdust samples were recovered (section 3.2.4.7) from the 2 remedially treated pole sections and from 2 pole sections which had been identically treated and maintained outside the soilbeds under the same conditions of temperature and relative humidity (section 3.2.2.4.). The sawdust samples were analysed for fluoride and chromium content (section 3.2.5.4.1.).

In the following presentation of results, pole sections 2 and 4 were those from model units TSS and TS respectively, while pole sections 1 and 3 were stored outside the soilbeds.

3.3.7.2. List of tables.

Table 3.3.7.1 shows the fluoride and chromium concentrations found 175 mm above and below the groundline (positions 1 and 3 respectively) and at the groundline itself (position 2) of remedially treated pole sections 2, 4, 1 and 3.

Table 3.3.7.2 indicates the moisture content of pole sections 2, 4, 1 and 3 at the groundline, before remedial treatment, and at 175 mm above and below the groundline after their respective exposures (section 3.3.7.1.).

Table 3.3.7.1. Mean percentage fluoride and chromium concentrations of wood samples from the groundline (2) and positions above and below the groundline (1, 3) of remedially treated pole sections, 2 and 4, exposed to the conditions of model units TSS and TS respectively and similarly treated pole sections, 1 and 3, maintained under cover outside the model units (standard deviations* and standard errors** in parenthesis for means of 2 and 6 respectively).

Pole No.	Sampling Position	Mean Fluoride Concentration (% w/w)*	Mean Chromium Concentration (% w/w)*	Mean Fluoride Concentration (% w/w)**	Mean Chromium Concentration (% w/w)**
2	1	0.8482 (0.0081)	0.2957 (0.0670)	0.7049	0.2362
(TSS)	2	0.4947 (0.0030)	0.2548 (0.0025)	(0.0688)	(0.0322)
	3	0.7717 (0.0578)	0.1580 (0.0817)		
4	1	0.5935 (0.0438)	0.2749 (0.0235)	0.6149	0.2756
(TS)	2	0.8561 (0.0787)	0.3670 (0.1121)	(0.0885)	(0.0403)
	3	0.3950 (0.1140)	0.1850 (0.0489)		
	1	0.5240 (0.0010)	0.2157 (0.0134)	0.5865	0.2844
1	2	0.7379 (0.1019)	0.3517 (0.0162)	(0.0528)	(0.0252)
	3	0.4978 (0.0611)	0.2857 (0.0061)		
	1	0.5583 (0.0107)	0.2183 (0.0012)	0.4440'	0.1729
3	2	0.3543 (0.0315)	0.1524 (0.0250)	(0.0494)	(0.0176)
	3	0.4194 (0.1697)	0.1480 (0.0493)		

Table 3.3.7.2. Moisture contents (%) of remedially treated pole sections 2 and 4, exposed to the conditions of model units TSS and TS respectively, and similarly treated pole sections 1 and 3, maintained under cover outside the model units.

Pole Number	Groundline Moisture Content (%) before Treatment	Moisture Content (%) 175 mm above/below the groundline after Exposure
2 (TSS)	12.5	19 / 24
4 (TS)	11	17 / 27
1	19	16 / 17
3	23	18 / 19

3.3.7.3. Fluoride and chromium concentrations in pole sections.

The mean fluoride concentrations of pole sections 2 and 4 were not significantly different (table 3.3.7.1). Similarly, the mean fluoride concentrations of pole sections 1 and 3, were not significantly different (table 3.3.7.1).

The mean chromium concentrations of pole sections 2 and 4 were not significantly different, though the mean chromium concentration of pole section 1 was significantly higher than that of pole section 3 (table 3.3.7.1).

Apart from the mean fluoride concentration in pole section 2 and the mean chromium concentration in pole section 4, which were significantly greater than the corresponding concentrations in pole section 3, $P = 0.012$ and 0.041 respectively, there were no significant differences between the mean fluoride or chromium concentrations, combined for all sample positions, of leached and unleached pole sections (table 3.3.7.1).

3.3.7.4. Moisture contents of pole sections.

Prior to remedial treatment with Rentex the moisture contents of pole sections 1 and 3, which were not to be exposed to the leaching conditions within the model units, were approximately double that of pole sections 2 and 4, which were to be exposed to the conditions within model units TSS and TS respectively (table 3.3.7.2). After their respective exposures, the moisture contents above and below the groundline of pole sections 1 and 3 were similar, and were lower than the earlier moisture contents recorded for these pole sections. In pole sections 2 and 4, the moisture contents below the groundline were markedly higher than those moisture contents above the groundline, which in turn were markedly higher than the moisture contents recorded for these pole sections prior to exposure and similar to the above and below groundline moisture contents of pole sections

1 and 3 (table 3.3.7.2). As expected therefore, the moisture contents of pole sections 2 and 4 displayed a marked increase during exposure to the conditions of the model units, while the moisture contents of pole sections 1 and 3 displayed a marked reduction.

3.4. DISCUSSION.

3.4.1. Introduction.

Discussion of the environmental effects associated with remedially treated timber, using the three model field units (section 3.3.), divides naturally into three parts. Section 3.4.2. examines the movement of the preservative components fluoride and chromium from remedially treated timber to the soil and soil drainage water, while section 3.4.3. examines the effects of these elements on plants and microbial activity in the model systems. The validity of the laboratory based model for use in an environmental impact assessment of treated timber will be evaluated, in section 3.4.4., by comparison of the findings from the model with those from field studies. Each part of the discussion opens with a statement of the main findings indicated by the relevant results section.

3.4.2. The movement of the preservative fluoride and chromium to the model environment.

3.4.2.1. The fluoride and chromium concentrations in pole sections after exposure to simulated field conditions.

The mean fluoride or chromium concentrations in treated pole sections, after exposure to the conditions in model units TSS and TS, were not significantly lower than those in similar pole sections unexposed to these conditions (section 3.3.7.3.). This showed that the extent of any reduction caused by exposure was masked by the variabilities associated with the sampling and analyses of the pole sections. Given this absence of a demonstrable loss of these elements from the pole sections in TSS and TS, the possibility that the particular soil conditions within these model units would have a significant bearing on the loss of these preservative components was slight. Hence, similar concentrations of fluoride and chromium were found in the pole sections which had been exposed to the conditions in model unit TSS and TS (section 3.3.7.3.).

These findings indicated that chemical analyses of the pole sections alone could not be used to establish the release of preservative fluoride and chromium to the model environments over the six month period of the experiment (table 3.2.2). This was in agreement with similar findings for individual pole sections exposed in the field over longer periods. For example, pole section 7 which was exposed at the Tealing field site for two years (table 2.3.7.2) and pole section 4 which was stored indoors over the same period (table 2.3.7.1) had each received seventy preservative injections, were of the same diameter, and contained very similar concentrations of fluoride and chromium. The similar preservative concentrations in these 2 pole sections were found in spite of the obvious movement of these preservative components from pole section 7 to the surrounding soil (table 2.3.8.1). However, a group of seven treated pole sections including pole section 7,

exposed at Tealing for two years, contained significantly lower mean concentrations of these elements, amounting to approximately half those in a similar group of seven, including pole section 4, stored indoors for the same period (section 2.3.7.3.).

Therefore, the apparent absence of substantial reductions in concentrations of fluoride and chromium in the pole sections exposed to the simulated field conditions in the model units, highlighted the general difficulty in identifying leaching effects in treated structures based solely on chemical analysis of a small number of such structures. Though the use of small numbers of treated pole sections was a necessary limitation of the model system under discussion, the experimental importance of this timber, in establishing the leaching of preservative constituents, was minimised by the inclusion of soil and soil water as additional model components. Chemical analysis of these components permitted independent identification of leached preservative constituents (see sections 3.4.2.2. - 3.4.2.5.).

However, the leaching of preservative constituents is dependant on the moisture regime imposed on the treated structure, and while the magnitude of the moisture contents in pole sections exposed to simulated field conditions (table 3.3.7.2) were comparable with those found in poles and pole sections under real field conditions (tables 2.3.3.1, 2.3.4.1, 2.3.4.2 and 2.3.7.4), the pattern of moisture contents found in the model pole sections were subtly different from those found in field pole sections. The conditions in the model units resulted in moisture contents below the groundline of the pole sections from model units TSS and TS, being approximately 26 and 60 % higher respectively than those above the groundline (table 3.3.7.2). However, in 6 pole sections from a group of 8 exposed to field conditions at the Tealing field site for up to 20 months, where a similar effect occurred, the mean difference was only approximately 9 % (table 2.3.3.1).

This greater disparity between moisture contents displayed by the model unit pole sections was understandable given that no simulated rainfall applications were made after day 55 of a trial which lasted for 190 days (table 3.2.2). Hence, for approximately 4.5

months these pole sections were entirely dependant on the soil moisture levels of their respective soilbeds to maintain timber moisture contents similar to those found under Scottish field conditions, where similar time periods without precipitation are unknown. These soilbed moisture levels alone were clearly not sufficient to maintain a difference between the moisture contents above and below the groundline of these pole sections, similar to that found under field conditions. This suggested that in future model unit operations where a sizeable proportion of the treated structure under study is not in soil contact, the rainfall regime should be maintained over the entire trial period.

3.4.2.2. The total fluoride and chromium contamination of the drainage waters collected from each model unit.

The quantities of fluoride found in the total leachates collected from model units CS, TSS and TS over all rainfall simulations, at 209.52, 208.57 and 238.44 mg respectively, were largely similar and were all much greater than the 38.02 mg applied to each model unit in the simulated rainfall (section 3.3.2.5.1. and table 3.3.2.4). Given the generally similar total volumes of leachate collected from each model unit (section 3.3.2.3.2 and table 3.3.2.1), the similarity between the total quantities of fluoride collected from each model unit initially suggested that the leachate fluoride concentrations were controlled by the solubility of a fluoride bearing product such as CaF_2. Though the calcium concentration was not recorded for the model unit leachates, Farrah *et al* (1985) demonstrated that the fluoride concentration in such a solution would be in the region of 7.6 ug/cm^3 which is consistent with the solubility products of CaF_2 as quoted by Aylward and Findlay (1971). This concentration is greatly in excess of those found in the leachates from any model unit (table 3.3.2.3) indicating that the magnitude of the fluoride concentrations in these leachates was a function of the magnitude of the soil fluoride concentration characteristic of the soil through which the leachates drained.

As the soil in model units CS and TS possessed identical physico/chemical characteristics (table 3.2.1) and similar initial fluoride concentrations (section 3.3.4.3.1.), the similar total quantities of fluoride found in the drainage waters of both these model units, indicated that any preservative fluoride leached from the remedially treated timber present in the TS soilbed added little to the normal background fluoride content of these drainage waters. Though the total leachates collected from model unit TS over all rainfall simulations did contain 14 % more fluoride than the corresponding leachates of CS (section 3.3.2.5.1.), the majority of this small increase was accounted for by the greater volume of total leachates collected from model unit TS over all simulations (section 3.3.2.3.2.). However, the similarity between between the the quantities of fluoride in the total leachates collected from model units CS and TSS (section 3.3.2.5.1.), the latter containing sand amended soil, indicated a different conclusion.

As sand amendment of the soil in model unit TSS (section 3.2.2.1.) resulted in its background or initial fluoride concentration being significantly lower than those of the soils in model units CS and TS (see section 3.4.2.5.1.), leachate waters from the TSS soil type, in the absence of a treated pole section, would be expected to contain much less fluoride than the corresponding waters of CS and TS. In addition, the pH of the sand amended soil was 5.45 compared to 6.15 for the CS/TS soil (table 3.2.1), due to a reduction in its H^+ ion buffering capacity via a diminution of the proportions of organic and inorganic soil colloids, ie. the organic matter (table 3.2.1) and the clay content respectively. As the solubility of soil fluoride generally increases below a soil pH of 5 and above pH 6 (Larsen and Widdowson,1971; Gilpin and Johnson, 1980), the mobility of background soil fluoride in the sand amended TSS soil would be lower, further restricting the fluoride content of its drainage waters. Therefore, the similar fluoride quantities in the total leachates from each model unit at each rainfall simulation and over all 9 simulations (section 3.3.2.5.1.) strongly indicated that the presence of remedially treated timber in the TSS soil type resulted in significant fluoride contamination of the drainage waters leaving this soil profile.

In contrast to the findings for fluoride, the quantity of total Cr in the total leachates collected from model units TSS and TS over all simulations, at 49.09 and 56.44 mg respectively were well above the background level of 4.12 mg found in the total leachates from CS, which did not contain much more than that in the simulated rainfall (section 3.3.2.6.1. and table 3.3.2.8).

Chromium (VI) was found in the leachates collected from model units TSS and TS, but was not found in SR or the leachates from model unit CS (section 3.3.2.7.1.). Chromium (VI), a very mobile species in soils (Bartlett and Kimble, 1976 b; Cary et al, 1977 b; Bloomfield and Pruden, 1980), is the sole form of Cr in Rentex (section 1.6.2.) and is rarely if ever found in uncontaminated field soils, where the Cr (III) species dominates (Bartlett and Kimble, 1976 a, b; Cary et al, 1977 b; Bartlett and James, 1988). The total Cr contents of the leachates from the model unit CS were therefore entirely dependant on Cr (III). As the mobility of the Cr (III) species in soils is restricted above pH 5.5 due to precipitation (James and Bartlett, 1983 a; Bartlett and James, 1988; Calder, 1988; M^{c}Grath and Smith, 1990), the near neutral leachates from the weakly acidic soil of model unit CS (tables 3.2.1 and 3.3.2.2 respectively), contained only a small total quantity of total Cr (section 3.3.2.6.1.). However, in the leachates from TSS and TS the highest percentages of total Cr were generally found in the form of Cr (VI) at those rainfall simulations where the greatest quantities of total Cr were found (section 3.3.2.7.2. and table 3.3.2.13). The total Cr contents of the leachates from model units TSS and TS were therefore predominantly dependant on the leaching of the Cr (VI) species from remedially treated timber.

However, though generally similar quantities of total Cr were collected from model units TSS and TS at each rainfall simulation (section 3.3.2.6.1.), much greater proportions of total Cr were found as Cr (VI) in the total leachates from TS at each simulation compared with those in the total leachates from TSS (section 3.3.2.7.2.). In consequence, while only 19.50 mg or approximately 40 % of the total Cr in leachates from TSS was in the form of Cr (VI), in the leachates from TS, this form of Cr accounted for 38.29 mg or

approximately 70 % of the total Cr content (section 3.3.2.7.2. and table 3.3.2.13). Therefore, while the dependance of the total Cr contents of these leachates on Cr (VI) was not in doubt, the importance of Cr (VI) in directly determining the total Cr content was less in those leachates collected from model unit TSS, and this is discussed further in section 3.4.2.3.2.

These findings for fluoride and chromium, indicate that while preservative chromium contamination of the leachates from model units TSS and TS certainly took place, fluoride contamination was evident only in the leachates from model unit TSS. However, the fluoride and chromium contents of the leachates collected from different positions within the soil profiles of each model unit provided more information regarding the movements of these elements in model unit drainage waters (see section 3.4.2.3.) and showed that leachates from model unit TS were subject to contamination by preservative fluoride lost from remedially treated timber (see section 3.4.2.3.1.).

3.4.2.3. The pattern of fluoride and chromium contamination in the leachates collected from different positions within the soil profiles of each model unit.

3.4.2.3.1. Fluoride.

For all the model units, CS, TSS and TS, drain position within the soilbed had a significant effect on leachate fluoride concentrations. However, comparisons of the fluoride concentrations in individual leachates within each model unit and between model units indicated that leached preservative fluoride increased the normal background fluoride concentrations of certain leachates recovered from model units TSS and TS. Sand amendment of the TSS soilbed facilitated the entry of higher concentrations of leached preservative fluoride into the drainage waters and by its effects on drainage ensured greater movement of preservative fluoride to the base of this soilbed.

For all the model units, significantly higher fluoride concentrations were found in the leachates from drains 6, 7 and 8, situated higher in the soilbed, compared to those found in leachates collected from drains 5 and 9 at the base of the soilbed (section 3.3.2.5.2.). In the absence of remedially treated timber in model unit CS, the soil was evidently the main source of the fluorides in the leachates collected from this model unit. Therefore significantly different fluoride concentrations would not normally be expected in any group of leachates from this model unit. However, the greater movement of waters to the base drains of all model units (section 3.3.2.3.3.) most probably included a portion of the simulated rainfall applied to each model unit which had avoided soil contact by traversing the space between the soilbed itself and the side of the tank containing it. These so called sidewall effects are well known from lysimeter studies and are thought to be more pronounced in relatively unstructured soilbeds (Bergstrom, 1992), such as were used in this study, which are more prone to shrinkage than undisturbed soil monoliths.

As the fluoride concentrations of the simulated rainfall applied to each model unit soilbed were much lower than those of the leachates collected (table 3.3.2.3), sidewall effects would encourage dilution of the leachates from the base drains 5 and 9 from each model unit. This would give rise to increased fluoride concentrations in leachates from the drains 6, 7 and 8 higher in the soil profile irrespective of pole section treatment. However, the mean fluoride concentrations found in leachates from drains 6, 7 and 8 of TSS, at 4.810, 1.431 and 1.319 ug/cm^3 respectively and from TS, at 1.444, 1.088 and 1.179 ug/cm^3 respectively were generally significantly greater than that found in the corresponding leachates from the combined drain 6,7,8 of CS at 0.940 ug/cm^3 (section 3.3.2.5.2.). These data clearly indicated that preservative fluoride contamination of the leachates from higher in the profile of model units TSS and TS had occurred. The fluoride concentrations in these leachates from TSS were typically significantly greater than those in the corresponding leachates from TS, and for both TSS and TS the fluoride concentrations decreased significantly as the distance between the drain and the treated timber increased (section 3.3.2.5.2.). In addition, given the expected lower background concentration of fluoride in

the leachates from model unit TSS (see section 3.4.2.2.), leachates collected from the base drains 5 and 9 of the TSS profile also appeared to contain elevated concentrations of fluoride (section 3.3.2.5.2.).

These findings indicated favoured entry of preservative fluoride into the leachates from model unit TSS. This was not unexpected, as regardless of whether the bulk of the fluoride adsorption capacity in these soils is provided by the free positive charges of aluminium ions on the broken edges of clay minerals or by the positive charges on the surfaces of amorphous aluminium and iron hydroxides (Pluger and Friedrich, 1971; Omueti and Jones, 1980; Farrah et al, 1985; Peek and Volk, 1985), the latter occurring predominantly as films or discrete particles on the surfaces of clay minerals (Russell, 1980), sand amendment effectively diluted this clay fraction in the TSS soil. Therefore, the capacity of the TSS soil to immobilise fluoride would be lower than that of the TS soil. Though the lower pH of the TSS soil (table 3.2.1) would tend to favour greater fluoride adsorption (Larsen and Widdowson,1971; Gilpin and Johnson, 1980), the generally greater fluoride concentrations in leachates from the TSS soil (section 3.3.2.5.2.) indicated that the availability of fluoride adsorption sites was the major determinant of preservative fluoride mobility in these soils. Greater fluoride concentrations were found in leachates from areas closest to the remedially treated timber in both TSS and TS, as irrespective of each soils adsorbing capacity, the shorter the contact time with the soil, the less depletion by adsorption. In addition, with increasing distance from the treated timber, dilution by uncontaminated waters which had not come in contact with the timber would increase, thereby lowering the leachate concentrations of preservative fluoride. These points were particularly important with regard to the quantitative distribution of leached preservative fluoride in the drainage waters of model units TSS and TS.

As sand amendment lowered the clay and organic matter content of the TSS soil profile (table 3.2.1), the surface area of soil particles for adsorption of water was reduced. This soil therefore had a lower water holding capacity than the CS/TS soil (table 3.2.1), resulting in a

lowering of impedance to water movement. This feature of the TSS soil profile, aided by blockages of higher drains (section 3.3.2.3.1.), ensured that less than 10 % of the total volume of leachates collected from model unit TSS was collected from these drains, whereas in model units CS and TS this volume was approximately 33 % (section 3.3.2.3.3.).

Therefore, though the soil conditions in model unit TS were less favourable to the mobility of leached preservative fluoride than the TSS soil (see earlier), the physical effect of these soil conditions on the movement of drainage waters ensured that the contaminated waters higher in the profile of model unit TS were removed before extended contact with the soil profile. Hence, the contaminated waters higher in the profile of TS (section 3.3.2.5.3.) represented firstly, a greater quantity of fluoride than the leachates collected from the base of this unit, secondly, a greater quantity of fluoride than the uncontaminated waters higher in the profile of CS, and lastly, a much greater quantity than the contaminated waters higher in the profile of TSS (section 3.3.2.5.3.).

In model unit TSS, the reverse situation applied with the soil conditions favouring less adsorption of leached preservative fluorides, but more extended contact with the soil profile, by encouraging a greater proportion of total leachate flow to the base of the profile (section 3.3.2.3.3.). Consequently, the contaminated drainage waters from the base drains of model unit TSS (section 3.3.2.5.2.), represented firstly a much greater quantity of fluoride than the contaminated waters from higher in the soil profile of this model unit and a much greater quantity of fluoride than the uncontaminated waters collected from the base drains of model units CS and TS (section 3.3.2.5.3.).

3.4.2.3.2. Chromium.

Only for model units TSS and TS did drain position within the soilbed have a significant effect on the total Cr concentrations in their respective leachates. Comparisons of the total

Cr concentrations in individual leachates within each model unit and between model units showed that leached preservative Cr (VI) increased the normal background total Cr concentrations of all but one of the leachates recovered from model units TSS and TS. Sand amendment of the TSS soilbed facilitated the movement of most of the leached preservative Cr (VI) to the base of this soilbed.

The mean total Cr concentration of the combined base drain leachates from model unit CS, at 0.016 ug/cm^3 was almost identical to that in the combined leachates from the drains higher in the profile at 0.014 ug/cm^3 (section 3.3.2.6.2. and table 3.3.2.9). There was therefore no evidence of sidewall effects (see section 3.4.2.3.1.) in this model unit. This is explained by the fact that the simulated rainfall entering model unit CS frequently contained greater quantities of total Cr than the total leachates leaving this model unit (section 3.3.2.6.1.), with the total Cr concentration of the former at several rainfall simulations being actually greater than that of the latter (table 3.3.2.7). Therefore, though sidewall effects occurred in model unit CS (see section 3.4.2.3.1.), these were obscured for total Cr as the simulated rainfall diluent of the base drain leachates was not much less concentrated than the CS leachates themselves.

In contrast, the much greater concentrations of total Cr in the separate leachates collected from model units TSS and TS, in comparison to those from the corresponding leachates from model unit CS and simulated rainfall (section 3.3.2.6.2., and table 3.3.2.9), would certainly have allowed sidewall effects to show up in these 2 model units, leading to dilution of the base drain leachates. However, in the absence of any substantial and interfering background contribution to the total Cr concentrations of these leachates by the soils (see section 3.4.2.3.), sidewall effects were of little significance in obscuring the patterns of preservative Cr contamination in the leachates from TSS and TS.

For model unit TSS the greatest mean total Cr concentrations, at 0.751 and 0.688 ug/cm^3, were found in leachates from the higher drain 6 in particular and the base drain 5,

both in close proximity to the treated timber (section 3.3.2.6.2. and table 3.3.2.9). Similarly, the highest mean total Cr concentrations of 1.720 and 0.264 ug/cm^3 were also found in these leachates from TS (section 3.3.2.6.2. and table 3.3.2.9). The concentrations of total Cr found in the remaining leachates of these 2 model units indicated preferential further movement of Cr to the drainage waters at the base of the TSS soil profile, while in TS the total Cr concentrations were more equitably spread between the higher and lower drainage waters of the soil profile (section 3.3.2.6.2. and table 3.3.2.9).

As Cr (VI) was found in the leachates from drains 5 and 6 of both these model units at most rainfall simulations where collections were made (section 3.3.2.7.1. and table 3.3.2.11), where it accounted for the greater proportion of the total Cr concentration present (section 3.3.2.7.1. and table 3.3.2.12), Cr (VI) was evidently the principal Cr species leached from the treated pole sections and was therefore the original source of the bulk of the total Cr in the drainage waters of model units TSS and TS (see also section 3.4.2.2.). In model unit TS, Cr (VI) was found progressively less frequently in leachates as the distance between the treated timber and the drains from which the leachates were collected increased, irrespective of drain depth within the profile (section 3.3.2.7.1. and table 3.3.2.11). In model unit TSS, though the occurrence of Cr (VI) in leachates from more distant drains was also less frequent, this was particularly evident in the leachates from those more distant drains, 7 and 8, situated higher in the soil profile, while the occurrence and persistence of Cr (VI) was more favoured in leachates from the most distant drain 9 at the base of the soil profile (section 3.3.2.7.1. and table 3.3.2.11). In addition, Cr (VI) tended to account for progressively smaller proportions of the total Cr concentration present in the leachates from both model units with increasing distance from the remedially treated timber (section 3.3.2.7.1.). This was particularly evident in leachates from drain 9, where the proportion of total Cr found as Cr (VI) was normally below 50 % (section 3.3.2.7.1. and table 3.3.2.12). Clearly Cr (VI) in the drainage waters of both model units was subject to increasing depletion with longer soil contact.

This depletion was indicative of Cr (VI) reduction to Cr (III) in the presence of soil organic matter (Bartlett and Kimble, 1976 b; Cary et al, 1977 b; Grove and Ellis, 1980 a; Bloomfield and Pruden, 1980; Bartlett and James, 1988), and was clearly superimposed on the respective drainage properties of each soil profile. In model unit TSS, these properties favoured greater vertical water movement to the base drains, whereas in TS a more equitable spread of drainage waters between upper and lower profile drains was found (see section 3.4.2.3.1. and section 3.3.2.3.3.). The distinctive distribution patterns of total Cr concentration in the leachates collected from model units TSS and TS (section 3.3.2.6.2. and table 3.3.2.9) were therefore due to the combination of these chemical and physical effects within each soilbed. The movement of leached Cr (VI) followed the paths of drainage flow characteristic to each soil profile and lower mean total Cr concentrations in more distant leachates were encouraged by enhanced reduction of Cr (VI) to less mobile Cr (III) and dilution by uncontaminated background waters.

Therefore, though the lower organic matter content, pH and cation exchange capacity of the sand amended TSS soil (table 3.2.1) would tend to maintain higher soluble concentrations of total chromium by restricting Cr (VI) reduction to relatively insoluble Cr (III) (Bartlett and Kimble, 1976 b), favouring slower precipitation of Cr (III) (Grove and Ellis, 1980), and reducing Cr (III) adsorption (Bloomfield and Pruden, 1980) respectively, only the total Cr concentrations of leachates from drains 5 and 9 of TSS were greater than those of TS (section 3.3.2.6.2. and table 3.3.2.9), where the aforementioned soil properties complimented the preferential drainage path in TSS. Similarly, though the soil conditions in TS favoured lower soluble concentrations of total Cr, the total Cr concentrations of leachates from drains 6, 7 and 8 of TS were greater than those of the corresponding leachates from TSS (section 3.3.2.6.2. and table 3.3.2.9) because the lateral movement of drainage waters to these drains in TSS was very poor in comparison to TS (section 3.3.2.3.3.).

The respective drainage characteristics of these soil profiles were also responsible for the unusual finding, that despite the TS soil type favouring greater reduction of leached Cr (VI) to Cr (III), by virtue of its greater organic matter content (table 3.2.1), 70 % of the total quantity of total Cr found in leachates from this model unit was in the form of Cr (VI), while in the leachates from TSS, only 40 % was in this form (see section 3.4.2.2.).

In model unit TSS, substantial total Cr concentrations were only found, in descending order of concentration, in leachates from drains 6, 5 and 9 (section 3.3.2.6.2. and table 3.3.2.9), reflecting the preferential distribution of Cr (VI) in the leachates of this unit (see earlier). In TS, the greatest concentrations of total Cr were also found in leachates from drains 6 and 5 (section 3.3.2.6.2. and table 3.3.2.9), though the characteristic distribution of Cr (VI) in the leachates of this model unit (see earlier), ensured that the total Cr concentration in the leachates from drain 9 of TS was almost the lowest found (table 3.3.2.9).

As the lower water holding capacity of the TSS soil type (table 3.2.1) exerted less vertical impedance to water movement (see section 3.4.3.1.) the bulk of the total volume of leachates collected from this model unit was recovered from the base drains 5 and 9 (section 3.3.2.3.3.). Partly due to the restricted egress of leachates from drain 5 (section 3.3.2.3.1.), the volume of leachates recovered from drains 5 and 9 amounted to approximately 5 and 88 % of the total recovered from TSS (table 3.3.2.1), while the frequent blockages of drain 6 of this model unit (section 3.3.2.3.1.) ensured that the total volume of leachates collected via this drain was approximately 0.1 % of the total recovered from this unit (table 3.3.2.1). In contrast, the volume of leachates collected from base drains 5 and 9 of TS amounted to 15 and 53 % of the total collected from this unit (table 3.3.2.1), and as drain 6 of this model unit was not subject to blockage, the volume of leachates collected via this drain was approximately 5 % of the total recovered (table 3.3.2.1).

Hence in model unit TSS, the quantity of total Cr recovered from drains 6 and 5, situated closest to the remedially treated timber, was approximately 4 and 24 % respectively of the total amount recovered from this model unit (section 3.3.2.6.3., tables 3.3.2.8 and 3.3.2.10), whereas in TS the quantity of total Cr recovered from these drains represented 38 and 25 % respectively of the totals (tables 3.3.2.8 and 3.3.2.10). Conversely, the quantities of total Cr recovered from the most distant drain 9 of TSS and TS represented 70 and 25 % respectively of these totals. Clearly then, in TSS the bulk of the total Cr recovered was collected in leachates at the greatest distance from the treated pole section where, for both model units, greater reduction of leached Cr (VI) took place (see earlier). In TS a similar proportion of the total Cr recovered was collected in close proximity to the pole section where reduction of Cr (VI) was lowest (see earlier).

Therefore, though the soil conditions in TSS favoured the maintenance of leached Cr (VI) in solution the drainage properties of this soil profile ensured that a greater proportion of preservative Cr (VI) leached was subjected to longer soil contact. Consequently a greater quantity of the total Cr (VI) leached into this model unit suffered reduction to Cr (III) (see table 3.3.2.14). In contrast, though the TS soil type favoured greater Cr (VI) reduction, a larger proportion of the Cr (VI) leached into this model unit was spared prolonged soil contact by collection in leachates from drains 5 and 6, which ensured that a much greater quantity of the total amount of Cr (VI) leached into this model unit was collected (see table 3.3.2.14). Hence, the total quantity of Cr (VI) recovered in the leachates of model unit TS was approximately twice that of TSS (section 3.3.2.7.2. and table 3.3.2.13).

However, despite the much greater proportions of total Cr found as Cr (VI) in the leachates of model unit TS and the greater mobility of Cr (VI) in soils in comparison to Cr (III) (Bartlett and Kimble, 1976 b; James and Bartlett, 1983 c; Calder, 1988), the quantity of total Cr collected from TS was essentially the same as that collected from TSS. This was possibly due to the physico/chemical properties of the TSS soil type which favoured greater Cr (III) mobility by slowing the precipitation and adsorption of this species (see earlier),

thereby balancing its greater reduction of Cr (VI). Alternatively, the solubility of Cr (III) in both TSS and TS may have been improved by complexation with organic acids or soil extracts of organic matter which have been shown to maintain Cr (III) in solution at pH values well in excess of those at which Cr (III) precipitation usually occurs (Bartlett and Kimble, 1976 a; James and Bartlett, 1983 a, b).

3.4.2.4. The maintenance of fluoride and chromium contamination of the leachates from model units TSS and TS.

The 50 % increase in the volume of SR applied to each model unit at the 4th rainfall simulation on day 28 resulted in a short lived flush of both fluoride and total Cr in the total leachates collected from model units TSS and TS (sections 3.3.2.5.1 and 3.3.2.6.1. and tables 3.3.2.4 and 3.3.2.8). Thereafter, the quantities of fluoride in the total leachates collected from these model units at each rainfall simulation were always in excess of those collected from each model unit prior to the increase in SR volume. Clearly this indicated that the rainfall regime did not deplete the preservative fluoride source in the remedially treated timber to such an extent that a decrease in the amount of fluoride leached from the timber occurred. However, the decrease in the quantities of total Cr in the total leachates from model units TSS and TS in particular, after the volume driven increase at or around the rainfall simulation on day 28 (section 3.3.2.6.1. and table 3.3.2.8), were also matched by striking falls in the quantities of Cr (VI) in these leachates (table 3.3.2.13). This suggested that a rapid depletion of the soluble Cr (VI) source of the total Cr in these leachates (see sections 3.4.2.2. and 3.4.2.3.2.) was taking place in both model units, by both leaching of Cr (VI) from the timber, and its adsorption and reduction to insoluble Cr (III) within the timber. These events were strongly corroborated by the inverted pyramid shape of Cr (VI) occurrence in the leachates of both these model units (table 3.3.2.11), showing that the presence of Cr (VI) was transitory in leachates collected at greater distances from the treated timber.

In addition, the complete absence of this Cr species from the watertables of both model units after a further 5 months in contact with the treated timber (section 3.3.2.7.3.), demonstrated that this strongly oxidising species, which was found in both watertables at the final rainfall simulation (see drain 5, table 3.3.2.11), was reduced to insoluble Cr (III) after prolonged contact with soil organic matter. Further confirmation of the transitory nature of Cr (VI) contamination was given by the substantial decreases in the original Cr (VI) concentrations of the fresh leachates, collected during the rainfall simulations, after an ageing period (section 3.3.2.7.3.). Though these losses of Cr (VI) were possibly due to conversion to Cr (III) via reduction by soluble organic matter, previous studies of Cr (VI) reduction in solution (Bloomfield and Pruden, 1980) indicate that the weakly acidic and alkaline pH of the leachates in the present study (section 3.3.2.4.) might have precluded this. However, Bloomfield and Pruden (1980) used highly refined solutions entirely free of particulates for their studies of Cr (VI) reduction, while both the fresh and aged soil leachate samples used in this experiment were 'dirty' until filtering just prior to each Cr (VI) measurement (section 3.2.5.1.5.). Therefore the losses of Cr (VI) anions in these leachates was most likely due to adsorption by suspended particulates prior to removal on filtering and/or reduction to Cr (III).

3.4.2.5. Soilbed deposition of preservative fluoride and chromium.

3.4.2.5.1. Background soil concentrations of fluoride and chromium in the model units.

The initial or background levels of fluoride and chromium in the soils of model units CS and TS, prior to pole section insertion, were not significantly different (section 3.3.5.3.1. and 3.3.5.4.1.). However, the sand amended soil of model unit TSS, generally contained significantly lower background levels of fluoride and chromium than that of CS and TS (section 3.3.5.3.1. and 3.3.5.4.1.). The lower fluoride and chromium concentrations of the

TSS soil were due to sand dilution of the major fluoride and chromium bearing clay and silt fractions of the original CS/TS soil (Gilpin and Johnson, 1980; Omueti and Jones, 1980; Peek and Volk, 1985; Calder, 1988).

These differences between the TSS and CS/TS soil types, were matched by the lower cation exchange capacity, organic matter content, water holding capacity and pH of the TSS soil (table 3.2.1) due to sand amendment (see section 3.4.2.2.). Hence, in this discussion, comparisons between soil fluoride and chromium concentrations of the CS and TS soil profiles are used to indicate the soil contamination associated with the presence of remedially treated timber (section 3.4.2.5.2.), while similar comparisons between the TSS and TS soil profiles are used to indicate the influence of soil type on the pattern of soil contamination (section 3.4.2.5.3.).

3.4.2.5.2. Fluoride and chromium soil contamination associated with remedially treated timber.

The presence of a remedially treated pole section in the soilbed of TS resulted in preservative fluoride and chromium soil contamination which was restricted to within 5 cm of the entire buried pole section surface.

At each sampling depth within the soilbed of model unit TS, the fluoride and chromium soil concentrations within 5 cm of the remedially treated pole section, which ranged from 631.30 - 973.80 ug/g and 148.99 - 217.61 ug/g respectively, were always significantly greater than those in soil more than 10 cm from the timber, where concentrations ranged from 242.70 - 383.30 ug/g and 73.60 - 122.30 ug/g for fluoride and chromium respectively (sections 3.3.5.3.2. and 3.3.5.4.2., and tables 3.3.5.1 and 3.3.5.2). The fluoride or chromium concentrations in soil at distances greater than 10 cm from the timber at each sampling depth in this soilbed were generally not significantly different (sections 3.3.5.3.2. and 3.3.5.4.2.). In this soilbed, the only consistent significant differences between the

fluoride or chromium concentrations of similar soil samples from different depths, occurred in soil within 5 cm of the treated timber where significant reductions in fluoride and chromium concentration were found with increasing sample depth (sections 3.3.5.3.2. and 3.3.5.4.2.).

This restricted pattern of preservative fluoride and chromium contamination in the soilbed of TS was confirmed by comparisons with corresponding samples from the CS soilbed, within which there were almost no significant differences for comparisons of fluoride or chromium soil concentrations (sections 3.3.5.3.2. and 3.3.5.4.2), which ranged between 295.44 - 390.40 ug/g and 58.52 - 79.20 ug/g respectively (tables 3.3.5.1 and 3.3.5.2).

The pattern of preservative fluoride and chromium deposition within the soilbed of TS was in broad agreement with the patterns of increased fluoride and chromium concentration found in the drainage waters collected from this soilbed. This was logical given the drainage waters obvious importance as a vehicle for the leaching and distribution of these preservative constituents. However, this bulk movement from the treated timber only occurred during the few hours of the rainfall simulations which were carried out on 9 occasions over a 40 day period (see table 3.2.2.). During this period and in the 4.5 months afterwards (table 3.2.2.) the diffusion of fluoride and chromium from the treated timber would also have been an important soil deposition process.

Therefore, fluoride and chromium contamination of soil within 5 cm of the remedially treated pole section in the TS soilbed was due to the higher leached concentrations to which the soils in this region were exposed (see section 3.4.2.3.) and the diffusion of these preservative constituents from the treated timber. The greater fluoride and chromium contamination found in soil samples within 5 cm of the pole section at shallower sampling depths of up to 30 cm compared to the corresponding samples from the base of this soilbed was obviously due to greater exposure. This probably occurred by virtue of the vertical face

of these shallower soil samples being continuous with the 35 cm deep remedially treated zone of the pole section, while that of the deeper soil sample was not. Preservative deposition in the deeper soil samples must have relied more heavily on leaching and secondary diffusion from above.

The occurrence of fluoride and chromium contamination in soil at distances greater than 10 cm from the pole section in this soilbed would have relied heavily on leaching during the rainfall simulations. However, the consistent absence of fluoride and chromium contamination in soil up to 30 cm from the surface, at greater distances from the treated timber, occurred in spite of the leachates from drains 6, 7 and 8 at a depth of 25 cm (see figures 3.2.1 and 3.2.2) generally containing fluoride and total chromium concentrations which were significantly greater than those of CS (section 3.4.2.3.). Evidently this soil was not greatly exposed to these higher concentrations. Therefore, the generally decreasing fluoride and chromium concentrations found in leachates from drains 6, 7 and 8 with increasing distance from the treated timber (see section 3.4.2.3.), were evidently due to dilution by progressively less contaminated drainage waters from above, and the concentrations of fluoride and chromium found in these leachates became progressively less representative of the soil solution from above.

There were also no consistent indications of preservative fluoride or chromium deposition at distances greater than 10 cm from the pole section at the base of the TS soilbed. The absence of fluoride soil contamination was understandable given that the leachate fluoride concentrations in this region were no greater than those background concentrations in the soilbed of CS (section 3.3.2.5.2.), as a probable consequence of soil adsorption at the pole surface and the efficient removal of the more highly concentrated leachates from the upper profile preventing their movement to the base of the TS soilbed (see section 3.4.2.3.1.).

However, the absence of significant chromium deposition at the base of the TS soilbed, at distances greater than 10 cm from the timber, was more difficult to understand given that the total chromium concentrations in the leachates collected here were consistently significantly greater than those background concentrations in the soilbed of CS (section 3.3.2.6.2.). This may indicate that the reduction of Cr (VI) to Cr (III) which was noted in these leachates (section 3.4.2.3.2.) was not accompanied by increased chromium deposition via adsorption and precipitation of the reduced species. This supports those indications that the Cr (III) species was complexed (section 3.4.2.3.2.), thereby preventing significant deposition and allowing it to be leached through the soilbed. Given this, and the fact that the more mobile Cr (VI) species accounted for the majority of total chromium in the leachates collected from drains 5 and 6 in close proximity to the treated pole section in TS, diffusion of Cr (VI) from the treated pole section in the 4.5 months after the rainfall simulations had ceased, was possibly the most important process for the deposition of preservative chromium in this soilbed.

3.4.2.5.3. The influence of soil type on fluoride and chromium soil contamination associated with remedially treated timber.

Preservative fluoride and chromium deposition within the sand amended soil profile of model unit TSS was similar to that found in TS (section 3.4.2.5.2.), with contamination generally restricted to within 5 cm of the treated timber. However, while the deposition of these elements was favoured at shallower depths within the TS profile (section 3.4.2.5.2.), in TSS, fluoride and chromium deposition was favoured at the base of the profile, with no significant chromium contamination occurring at shallower depths. Sand amendment of the TSS soilbed ensured that the soil concentrations of fluoride and chromium from contaminated areas of this soilbed were not greater than corresponding soil concentrations from contaminated areas of the TS soilbed.

At each sampling depth within the soil profile of model unit TSS fluoride concentrations within 5 cm of the treated pole section, which ranged from 385.00 - 682.45 ug/g, were always significantly greater than those found in soil at distances greater than 10 cm from the timber, which ranged from 148.10 - 309.90 ug/g (section 3.3.5.3.2. and table 3.3.5.1). The fluoride concentrations in this soil at greater distances from the timber at each sampling depth in this soilbed were generally not significantly different (section 3.3.5.3.2.). Similar comparisons for chromium indicated that, though concentrations within 5 cm of the pole section were noticeably higher than those in more distant samples at the shallowest sampling depth, only at the base of this soilbed did soil concentrations decrease significantly with increasing distance from the pole section, ranging from 170.80 ug/g within 5 cm of the pole section to 80.90 ug/g at 50 cm (section 3.3.5.4.2. and table 3.3.5.2). The concentrations of fluoride and chromium in the soil samples from the base of this soilbed were consistently significantly greater than those in corresponding samples from the shallower sampling depths of this soilbed (sections 3.3.5.3.2. and 3.3.5.4.2.).

Therefore, despite the soil within 5 cm of the pole section at the base of the TSS soilbed suffering the same disadvantage as that in TS, in terms of its lower direct exposure to the treated zone of the pole section (section 3.4.2.5.2.), fluoride and chromium deposition was favoured here in the TSS soilbed, while in TS, deposition was favoured at shallower depth (section 3.4.2.5.2.). However, as the unamended field soil base of each soilbed (section 3.2.2.1.) accounted for almost half the volume of the soil samples from the base of each soil profile, in the sand amended soilbed of TSS these soil samples will have had a fluoride adsorption capacity and initial fluoride concentration which was greater than those of the shallower completely amended samples (see sections 3.4.2.3.1. and 3.4.2.5.1.). Consequently, though the shallower soil samples within 5 cm of the pole section in TSS were undoubtedly exposed to greater concentrations of preservative fluoride they possessed a lower adsorption capacity, thereby facilitating deeper leaching of fluoride for deposition in the soil of higher adsorption capacity and higher initial fluoride concentration at the base of the pole section.

Similarly, preservative chromium was mostly leached in the form of the mobile Cr (VI) anion (section 3.4.2.3.2.), and the deposition of chromium added to soil in this form is discouraged by the lower anion adsorption capacity, lower organic matter content and lower cation exchange capacity which characterised the sand amended TSS soil type (section 3.4.2.3.2.). Chromium (VI) would therefore tend to remain in solution for deeper deposition where the more unamended nature of the lower soil samples in the TSS soilbed would have provided a greater anion adsorption capacity, organic matter content and cation exchange capacity in comparison to the shallower soil samples in this model unit.

While this explains the greater deposition of preservative fluoride and chromium in the soil within 5 cm of the pole section at the base of the TSS soilbed, it does not explain why there was an absence of chromium contamination within 5 cm of the pole section at shallower depths in this model unit, when preservative fluoride deposition was found and there was both chromium and fluoride contamination within 5 cm of the pole section at shallower depths in the TS model unit (section 3.4.2.5.2.). This difference between the TSS and TS soilbeds in relation to the deposition of fluoride and chromium may have been due to a difference between these preservative components in terms of the relative importance of diffusion to the deposition process.

Though the preservative fluoride source in the treated timber did not apparently suffer extensive depletion during the period of rainfall simulations, severe depletion of the Cr (VI) source by leaching and reduction to Cr (III) in the timber was indicated (section 3.4.2.4.). Therefore, while the diffusion of fluoride from the treated timber would have continued during the 4.5 months after the rainfall simulations had ceased, the diffusion of chromium during this period was probably restricted by the fixation of the now predominant Cr (III) species in the timber (see section 2.4.3.1.). Hence, while the increased soil concentrations of fluoride at shallower sampling depths within 5 cm the treated pole sections in both TSS and TS, were probably due to the combined leaching and diffusion of preservative fluoride, the soil concentrations of chromium in this area would have been more dependant on the

leaching of preservative chromium during the rainfall simulations. In which case, the effect of the TSS soil type in comparison to the TS soil type in reducing chromium deposition at shallower sampling depths by reducing the adsorption of leached chromium, would have been much greater than its effect in reducing fluoride deposition by reducing the adsorption of leached fluoride. This effect would be even more pronounced given the indications that diffusion may have been the most important process with regard to the deposition of preservative chromium (section 3.4.2.5.2.).

Irrespective of these differences between preservative fluoride and chromium movement from the treated timber, the soil from shallower sampling depths of up to 30 cm within 5 cm of the remedially treated timber in both TSS and TS, were subjected to similar fluoride and chromium exposure. In spite of this, the fluoride and chromium concentrations in this soil from the TS soilbed were significantly greater (sections 3.3.5.3.2. and 3.3.5.4.2.) due to this soils significantly greater initial fluoride and chromium concentration (see section 3.4.2.5.1.), greater fluoride adsorption capacity and greater capacity to adsorb chromium (see section 3.4.2.3.).

In contrast, the soil from within 5 cm of the pole section at the base of the TSS soilbed was subjected to greater exposure to leached preservative fluoride and chromium than the corresponding sample from TS. However, despite this, the fluoride and chromium concentrations in this soil from the TSS soilbed was only similar to that in the corresponding TS sample (sections 3.3.5.3.2. and 3.3.5.4.2.). This occurred because the greater adsorption capacity of the TS soil (see earlier) encouraged greater adsorption of fluoride and chromium to add to its greater initial fluoride and chromium soil concentration (see section 3.4.2.5.1.). Though the inclusion of unamended field soil in the samples from the base of each soilbed (see earlier) will have resulted in a greater parity between the initial fluoride/chromium concentrations and fluoride/chromium adsorption capacities of these soil samples from each model unit, than between those of the shallower soilbed samples (see section 3.4.2.3.), these differences were clearly not removed.

The absence of significant fluoride and chromium contamination in soil at distances greater than 10 cm from the pole section at depths up to 30 cm in the TSS soilbed, was generally confirmed by the significantly greater fluoride and chromium concentrations which were consistently found in the corresponding uncontaminated soil samples from the TS soilbed (sections 3.3.5.3.2. and 3.3.5.4.2.). Hence, as indicated for the TS soilbed (section 3.4.2.5.2.), the elevated fluoride and chromium concentrations in the leachates from drains 6, 7 and 8 were not representative of the soil solution within this area. Clearly, the absence of contamination in this soilbed area of both TSS and TS ensured that the significantly greater initial fluoride and chromium concentration of the TS soil (section 3.4.2.5.1.), was generally maintained.

However, significantly greater concentrations of fluoride and chromium were consistently found in soil samples at distances greater than 10 cm from the pole section at the base of the TSS soilbed compared to corresponding shallower soil samples from this profile (sections 3.3.5.3.2. and 3.3.5.4.2.). These greater concentrations at the base of the TSS soilbed were confirmed by the consistent absence of significant differences between these concentrations and those in corresponding samples from the TS soilbed (sections 3.3.5.3.2. and 3.3.5.4.2.). These findings probably did not indicate significant preservative contamination of the TSS base samples as the more unamended nature of soil from this area of the TSS soilbed would have provided these soil samples with higher concentrations of fluoride and chromium than in the rest of this soilbed, and would have encouraged similar concentrations to those in corresponding samples from the TS soilbed in any case. As the leachates collected from this area of the TSS soilbed contained consistently higher concentrations of fluoride and chromium than those in the corresponding leachates from CS and TS (section 3.4.2.3.), the fluoride and chromium leached from the treated timber during the rainfall simulations did not contribute greatly to the soil deposition of these elements and was apparently freely lost from the soilbeds. This indicates that the significant fluoride and chromium soil contamination associated with the remedially treated pole sections in TSS and TS, was primarily due to diffusion, and hence it did not extend to distances greater

than 10 cm from the timber.

3.4.3. The effects of leached preservative fluoride and chromium on plants and microbial activity in the model unit environment.

3.4.3.1. Perennial ryegrass swards.

3.4.3.1.1. Dry matter yield and foliar fluoride and chromium concentrations in consecutively grown grass swards.

Significant dry matter yield reductions were found in consecutive swards of ryegrass cultivated adjacent to remedially treated timber in model units TSS and TS. The yield reductions however, extended to no more than 10 cm from the treated timber. Though individual samples of ryegrass in close proximity to the treated timbers displayed excessive levels of foliar fluoride and chromium indicative of the accumulation of leached preservative constituents, yield reductions were not directly related to increased foliar concentrations of these elements.

Comparisons between the dry matter yields of corresponding first sward grass samples from the soilbeds of CS, TSS and TS, showed that yields within area A/B/C of TSS and TS, at 0.1415 and 0.1887 g/50 cm^2, were significantly lower than that of CS, at 0.2546 g/50 cm^2 (section 3.3.3.3.1. and table 3.3.3.1). This area was equivalent to 150 cm^2 and extended to 5 cm downslope of the pole section in each soilbed (figure 3.2.6). Similar comparisons between corresponding second sward grass samples from these soilbeds showed that the area over which these significant reductions in grass yield occurred had extended to include area D/E/F, where yields of 0.1714, 0.1504, 0.2086 g/50 cm^2 were found for TSS, TS and CS respectively (section 3.3.3.3.2. and table 3.3.3.1). The combined area of A/B/C and D/E/F was 300 cm^2 extending to 10 cm from the pole sections (figure 3.2.6).

Despite these findings, grass yields found in different sample areas within either sward in both TSS and TS, were predominantly similar (section 3.3.3.3.). However, in the CS soilbed, the yield of the first sward grass samples from area A/B/C and the yield of the second sward samples from areas A/B/C and D/E/F were significantly greater than the more distant grass samples from their respective swards (section 3.3.3.3.).

These findings suggested that the progressive reductions in grass yields adjacent to treated timber in model units TSS and TS, by comparison with identical areas of grass sward in CS, was not due to a simple reduction in the normal growth of grass in the former model units. Instead, these yield reductions appeared to be caused by disruption of a progressive effect which favoured increased grass growth adjacent to the control pole section in CS. These favourable growth conditions may have been due to the leaching of soluble nutrients from the timber (Smith, 1980), as increased soil microbial activity was also found in the soil adjacent to the pole section in this model unit (see section 3.4.3.3.). The effect of remedial treatment was therefore to eliminate these enhanced growth conditions adjacent to the pole section. Hence, grass yields in areas immediately adjacent to the treated timbers in model units TSS and TS were generally not significantly different from yields in the rest of the sward.

The presence of preservative fluoride and chromium as generally significant contaminants of the soil and its drainage waters in the rhizosphere area of model units TSS and TS where these grass yield reductions occurred (sections 3.4.2.3. and 3.4.2.5.), strongly implicated these leached preservative consituents as the causal agents. However, the mean foliar fluoride and chromium contents of the first or second sward grass samples from corresponding areas of CS, TSS and TS were not significantly different (section 3.3.3.4.). Nevertheless, much higher non-significant mean foliar fluoride and chromium concentrations were generally found in grass samples from model units TS and TSS, in particular, with mean concentrations of foliar fluoride and chromium reaching levels of 51.80 and 96.10 ug/g respectively in first sward grass samples within 5 cm of the pole

section in TSS compared to 4.24 and 15.80 ug/g respectively in corresponding samples from CS (table 3.3.3.1).

These findings clearly showed that enhanced plant uptake of fluoride and chromium was taking place at a number of positions within these swards (table 3.3.3.1). These indications of enhanced uptake in sward samples from TSS and TS were confirmed by the much higher foliar fluoride and chromium concentrations found in first and second sward grasses recovered from within 5 cm of the back and sides of the treated pole sections compared to those in the corresponding samples from the CS soilbed (section 3.3.3.5.). This was particularly the case with regard to the foliar chromium concentrations in the grasses from TSS which, at 175.03 and 294.09 ug/g for first and second sward samples respectively, were greatly in excess of corresponding values for samples from CS and TS (table 3.3.2.2). These foliar chromium concentrations were also clearly in excess of the soil concentration of 62.09 ug/g in this area (table 3.3.5.2.), undoubtedly indicating that root absorption and shoot translocation of chromium from the soil solution adjacent to this pole section had taken place.

These findings were in line with the favourable conditions found in the TSS soil type for the maintenance of leached preservative fluoride and chromium (VI) in solution (sections 3.4.2.3.1. and 3.4.2.3.2.), as fluoride uptake by plants is directly related to the amount of soluble fluoride available in the growing medium (Leone et al, 1948; Hara et al, 1977; Singh et al, 1979 b) and plants supplied with Cr (VI) at the root, consistently possess higher concentrations of foliar chromium than those supplied with the reduced Cr (III) species (Skeffington et al, 1976; Cary et al, 1977 a; Lahouti and Peterson, 1979; Ramachandran et al, 1980; M^{c}Grath, 1982), due to the enhanced mobility of the former species (Skeffington et al, 1976; Lahouti and Peterson, 1979).

3.4.3.1.2. Sward densities of selected second sward grass samples.

Sward density reductions in the second ryegrass swards of model units TSS and TS were most pronounced within 5 cm of the treated timbers. However, the area of sward within 10 cm of the remedially treated timber in TSS and TS each displayed a significant reduction in sward density by comparison with the corresponding sward in model unit CS. These sward density reductions were responsible for the yield reductions noted in this area of model units TSS and TS (section 3.4.3.1.1.).

While a number of significant differences were found between the mean leaf numbers in sample positions 1 - 6 within the grass sward in each soilbed, only in model units TSS and TS in particular, were grass samples from positions 3 and 4, subject to a reduction in sward density (section 3.3.3.6.). These sward density reductions covered an area of 50 cm^2 extending to 5 cm directly downslope of the remedially treated pole sections (figure 3.2.6). Further comparisons of sward density between sample positions within TSS and TS indicated that these reductions did not extend any further from the treated pole sections (section 3.3.3.6. and figure 3.2.6). These findings were corroborated by comparisons between the sward densities in corresponding sample positions of different model units, where only those of grass samples from positions 3 and 4 of TSS and TS were always significantly lower than those from CS (section 3.3.2.6.).

Though sward density reductions were therefore particularly evident closest to the treated timber, the mean leaf numbers over all sample positions 1 - 12 of model units TSS and TS, at 7.18 and 7.34 respectively, were significantly lower than CS at 9.24 (section 3.3.3.6.). Hence, sward density was significantly reduced over this entire sampling area corresponding to 300 cm^3 up to 10 cm downslope of the remedially treated pole sections in model units TSS and TS (figure 3.2.6). This had obvious implications with regard to the significant yield reductions which were found in the same area of these swards in model units TSS and TS (see section 3.4.3.1.1.).

While the dry matter yield of swards in this area model units TSS and TS amounted to approximately 72 and 69 % respectively of that in CS (table 3.3.3.1), the sward density in this area of model units TSS and TS was approximately 78 and 79 % repectively of that in CS (table 3.3.3.3). These findings indicated that the significant reductions in dry matter yield of grass in close proximity to the remedially treated pole sections (section 3.4.3.1.1.) were largely due to a fall in sward density. This was to be expected considering that these grass swards had been established for barely 2 weeks and that the principal phytotoxic effects of fluoride and chromium, ie. interference with the root uptake of essential elements (Hewitt, 1953; Hunter and Vergnano, 1953; Turner and Rust, 1971) and inhibition of photosynthesis (M^{c}Laughlin and Barnes, 1975; Parry et al, 1984) respectively, would be enhanced in these early growth stages when nutrient availability and photosynthetic efficiency are vital pre-requisites for plant survival. Therefore, the reduced yields of grass found within 10 cm of the pole sections in TSS and TS were caused primarily by a reduction in sward density due to reduced seedling survival.

By implication, reduced dry matter yield (section 3.4.3.1.1.) due to reduced plant size via the toxic effects of increased foliar fluoride or chromium concentrations (Hansen et al, 1958; Mortvedt and Giordano, 1975; Singh et al, 1979 a, b; M^{c}Grath, 1982) was of minor importance in the affected swards. This possibly accounts for the difficulty in finding significantly higher foliar fluoride and chromium concentrations in grass samples of significantly reduced yield (section 3.4.3.1.1.).

3.4.3.2. Rye plant canopies.

3.4.3.2.1. Rye seedling viability.

Rye seedling viability was not significantly affected by the presence of remedially treated timber in model units TSS and TS.

The differences in Rye seedling viability between model units were within the bounds of natural variation (section 3.3.4.3.), probably brought about by inter-plant competition for light and nutrients within the crop. Therefore the generally significant increases in soil concentrations of fluoride and chromium, found in the rhizosphere area in close proximity to the treated pole sections in model units TSS and TS (section 3.4.2.5.), in the region occupied by seeds planted in rows 1 - 4 in particular (figure 3.2.7), had no measureable effect on the emergence of this crop. Unlike, the grass swards, Rye was sown after the period of rainfall simulations (table 3.2.2), when increased concentrations of fluoride and chromium appeared in the drainage waters. Soil adsorption of these leached preservative constituents in close proximity to the treated pole sections (section 3.4.2.5.) probably discouraged uptake by the Rye seedlings and hence no phytotoxic effects were recorded, to match those found in the grass swards (section 3.4.3.1.).

3.4.3.2.2. Plant growth within the crop canopy.

The growth of rye plants was not adversely affected by cultivation adjacent to the remedially treated pole sections in model units TSS and TS. The soil conditions within the sand amended soilbed of model unit TSS, supported the plant canopy of greatest plant density and plant size, in spite of the presence of remedially treated timber. However, root development within each crop canopy was restricted due to the soilbed conditions.

Comparisons of a range of growth measurements, consisting of canopy heights, crop densities, leaf production and longevity, dry weight yields and rooting depths, between sectors within the crop of each model unit indicated that any significant growth effects found in close proximity to the pole section in each model unit were no more consistent than similar random effects which occurred throughout each crop canopy (section 3.3.4.4.). Hence, any effects on the growth of rye plants in any of the 3 soilbeds, due to the presence of pole sections, irrespective of preservative treatment, were not significant. This was the case even in sampling sectors within 10 cm of the treated timbers in model units TSS and TS where concentrations of fluoride and chromium in the rhizosphere soil were consistently in excess of background levels (section 3.4.2.5.). However, as the survival of rye seedlings in this area was not affected by these soil contaminants (section 3.4.3.2.1.), and as these soil concentrations were not widely spread across the slope of each soilbed (section 3.4.2.5.) the absence of any negative effects in more mature plants was perhaps to be expected.

Where significant differences were found between crop growth parameters in corresponding sectors of different model units, these always favoured the plants grown in model unit TSS (section 3.3.4.4.). Similarly, the mean leaf number, total viable leaf length, shoot dry weight and root dry weight of plants over all sectors of model unit TSS, at 6.55, 111.28 cm, 51.8 mg and 2.1 mg respectively were significantly greater than those of both CS at 6.14, 82.58 cm, 33.8 mg and 1.6 mg respectively, and TS, at 5.85, 85.61 cm, 38.3 mg, and 1.5 mg respectively (section 3.3.4.4., and tables 3.3.4.3 and 3.3.4.4). The crop canopy of model unit TSS also contained 30 % and 20 % more individual Rye plants than CS and TS respectively (section 3.3.4.4.2.).

These findings clearly confirmed the absence of a significant negative effect on Rye plant growth due to the presence of remedially treated timber, and indicated the superiority of the Rye crop grown on the sand amended soilbed of model unit TSS. The significantly greater root dry weight of plants in the soilbed of TSS (section 3.3.4.4.3.) probably occurred in response to the lower water holding capacity of this sand amended soil (table

3.2.1) allied to a possibly improved soil texture for root expansion. As all the soilbeds were periodically sprayed with a standard NPK plant fertiliser (section 3.2.3.1.) the plants in the sand amended soilbed were therefore at a distinct advantage in terms of root surface area for nutrient uptake. In addition, the lower ion exchange capacity of this soil (section 3.4.2.3.), would enhance the availability of these nutrients to the crop by enhancing their availability in the soil solution.

However, the mean root dry weight/mean shoot dry weight ratios for plants in each soilbed, at 0.046, 0.041 and 0.039, for CS, TSS and TS respectively (see table 3.3.4.4), were below a range of 0.30 - 0.83 for cereals and grasses grown under a variety of conditions (Russell, 1977). As the lower ratios of this range were due to high levels of nitrogen and phosphate supplied to the root, which removes the impetus for the development of large root systems, it is unlikely that the addition of fertiliser alone could have resulted in the much lower root/shoot ratios displayed by the rye plants in all model units. It is probable that restricted root development due to fertiliser application was further retarded by the moist soil conditions maintained in each soilbed (section 3.2.3.4.), which provided ample nutrients and water at shallow depth.

3.4.3.3. Dehydrogenase activity in the topsoil of model units CS, TSS and TS.

3.4.3.3.1. Dehydrogenase activity in unsupplemented soil as influenced by the presence of remedially treated timber.

Over a measurement period of 4 weeks during which the dehydrogenase activity of unsupplemented soil from each model units was followed, the activity of soil from within 5 cm of the remedially treated pole sections in TSS and TS especially, samples 1 and 3, was consistently significantly lower than that in more distant samples from these model units (section 3.3.6.3.). These reduced activities were confirmed by comparisons with the

dehydrogenase activities of corresponding unsupplemented soil samples from model unit CS, where a tendency for greater dehydrogenase activity within 5 cm of the non-remedially treated pole section was found (section 3.3.6.3.).

Clearly the reductions in soil dehydrogenase activity in close proximity to the treated pole sections in TSS and TS were in response to the generally elevated concentrations of preservative fluoride and chromium found in these soils (sections 3.4.2.5.2. and 3.4.2.5.3.). These findings are in line with those of a number of studies in which reduced dehydrogenase activities were found in soils contaminated with heavy metals and pesticides (Ruhling and Tyler, 1973; Doelman and Haanstra, 1979; Schinner *et al*, 1980; Davies and Greaves, 1981; Rogers and Li, 1985; Bitton and Koopman, 1986; Chander and Brookes, 1991 b; Hainey, 1992). Dehydrogenase enzymes are produced by soil organisms to catalyse the transfer of hydrogen from organic substrates to molecular oxygen to form water (Ruhling and Tyler, 1973; Chander and Brookes, 1991 a) though measurements of this activity in soil do not give absolute levels of microbial respiration (Howard, 1972; Benefield *et al*, 1977). However, measurements of enzyme inactivation are regarded as one of the most relevant techniques for determining the effects of pesticides and heavy metals on soil microflora (US EPA, 1978; Forstner, 1988) and the findings in this study clearly demonstrated its usefulness in establishing a general deleterious effect on microbial activity in soil adjacent to remedially treated timber.

This was in sharp contrast to the findings for CS, where dehydrogenase activity was apparently encouraged in soil in close proximity to the pole section. Though the aged pole section in CS was originally creosoted, in the leachates collected from this model unit, there was no evidence of the oily contaminants which were always found in moist soil around creosoted field poles in this study, and it is reasonable to assume that this pole section essentially behaved as untreated timber. Increased dehydrogenase activity has been found previously in soil adjacent to unpreserved wood (Mowe, 1983; Green, 1988; Hainey, 1992) and may be due to enhanced microbial germination via the diffusion of soluble wood

nutrients to the soil (Smith, 1980), stimulation of fungal growth by volatiles released from the wood (Mowe and King, 1981), and/or the maintenance of a biotic connection between the soil and wood during the decay process (King et al, 1980). While, the increased activity in soil adjacent to the pole section in CS, was probably due to microbial responses to these events, it is clear that the preservative soil contamination adjacent to the treated timber in TSS and TS, prevented such responses.

3.4.3.3.2. Dehydrogenase activity in unsupplemented soil as influenced by soil type.

The dehydrogenase activities of unsupplemented soil samples from the sand amended TSS soilbed were invariably significantly lower than those of corresponding soil samples from the unamended CS and TS soilbeds (section 3.3.6.3.). This general negative effect of sand amendment on soil activity was probably related to the reduced organic matter content of this soil type, at 18.8 g/kg compared to 33.3 g/kg for the CS/TS soil type (table 3.2.1), due to dilution of this soil fraction in the TSS soil (section 3.2.2.1.). As the organic matter content of the soil forms the primary source of energy for the microbial population (Russell, 1980) reduced dehydrogenase activity would be expected in the TSS soil.

The cumulative negative effect of preservative soil contamination and lower organic matter content on microbial activity within the TSS soilbed was indicated by the generally lower dehydrogenase activities found in soil samples from within 5 cm of the treated pole section in TSS compared to corresponding samples from TS (section 3.3.6.3. and tables 3.3.6.1 and 3.3.6.3), despite the significantly higher topsoil concentrations of fluoride and chromium found within 5 cm of the pole section in TS compared to TSS (sections 3.3.5.3.2. and 3.3.5.4.2.).

3.4.3.3.3. Dehydrogenase activity in soil as influenced by rye meal supplementation.

Supplementaton of the organic matter content of the soils with rye meal resulted in a predictable increase in the levels of dehydrogenase activity in all soil samples (section 3.3.6.4.), due to an increase in microbial biomass. However, these increased activities did not obscure the effects of pole treatment or sand amendment in the unsupplemented soils of CS, TSS and TS (sections 3.4.3.3.1. and 3.4.3.3.2.). Indeed, whereas the effects of pole treatment in unsupplemented soils (section 3.4.3.3.1.) were more clearly indicated for comparisons of mean values combined over all 4 measurements (section 3.3.6.3.), supplementation highlighted these effects on a week by week basis (section 3.3.6.4.).

These findings showed that though the impaired microbial activity in soil adjacent to the remedially treated timbers in TSS and TS was capable of stimulation by increasing the available supply of organic matter, it still remained lower than the background levels in soil at greater distance from the poles. These findings are in accord with those of Chander and Brookes (1991 b) who demonstrated that while the microbial biomass in soils of both low and high metal concentrations was increased by maize and glucose supplementation, the lower biomass of the original unamended high metal soil was maintained, due to less efficient utilization of substrates for biomass synthesis. This would explain why rye supplementation magnified the negative and positive activity effects which were found in the unsupplemented samples from each model unit at each week of measurement (compare sections 3.3.6.3. and 3.3.6.4.). These findings indicate the usefulness of soil supplements in highlighting and confirming the effects of contamination on soil microflora.

3.4.4. Evaluation of the physical field model findings with respect to the environmental impact of remedially treated timber in the field.

The physical field model allowed an accurate assessment of preservative fluoride and chromium contamination in soil and drainage water adjacent to remedially treated timber sections (sections 3.4.2.2. - 3.4.2.5.), and permitted the effects of this contamination, on plant growth and microbial activity, to be measured (section 3.4.3.).

The degree of preservative fluoride and chromium contamination in the drainage waters adjacent to remedially treated pole sections (sections 3.4.2.2. and 3.4.2.3.) indicated that the loss of these preservative constituents would not represent a serious threat to groundwater supplies. Similarly, though the movement of fluoride and chromium from the treated timbers resulted in significant soil contamination, this did not extend to distances greater than 10 cm from the treated timber (section 3.4.2.5.). Hence, deleterious effects on plant growth (section 3.4.3.1.) and microbial activity (section 3.4.3.3.) were restricted to this area. The accuracy of these findings with respect to the environmental impact of remedially treated timber in the field depended firstly, on the treated timber in the model units containing preservative concentrations which were comparable with those in the field, secondly, the extent to which preservative contamination of the model environment was comparable with that found in the field, and lastly, the similarity between the conditions in the control model unit CS and background environmental conditions in the field.

With regard to the treated timber, the mean fluoride concentrations, within the remedially treated pole sections exposed to the conditions in model units TSS and TS, at 0.7049 and 0.6149 % w/w respectively (table 3.3.7.1) were above the mean value of 0.3464 % w/w found for seven remedially treated pole sections exposed to the Tealing field site for 2 years, but were below the mean value of 0.7458 % w/w in seven treated field pole sections which had remained covered for 2 years after treatment (table 2.3.7.5). Similarly, the mean chromium concentrations, within the treated pole sections exposed in model units

TSS and TS, at 0.2362 and 0.2756 % w/w respectively (table 3.3.7.1) were above the mean value of 0.1980 % w/w in treated pole sections exposed for 2 years, but were below the mean value of 0.309 % w/w in pole sections which had remained covered for 2 years (table 2.3.7.5). As the conditions within the model units were intended to represent approximately 6 months field exposure, these findings indicated that the treated sections used in the model units were comparable with those used in the field.

Similarly, the mean concentrations of preservative fluoride and chromium in soil within 5 cm of the pole section in the unamended sandy loam soil of model unit TS, which ranged from 631.30 - 973.80 ug/g and 148.99 - 217.61 ug/g respectively throughout a depth of about 50 cm (tables 3.3.5.1 and 3.3.5.2), agreed well with with the mean fluoride and chromium concentrations, of 1033.80 and 201.34 ug/g, found over a depth of 60 cm in the sandy and sandy clay loam soils within 6 cm of fourteen distribution poles 6 months after remedial treatment at the Glenclova field site (table 2.3.6.5). In contrast to the limited contamination around the pole section in model unit TS, lower levels of fluoride and chromium contamination were found up to 25 cm from these field poles, 6 months after remedial treatment (section 2.3.6.2.). However, these field concentrations were always significantly lower than those in soil within 6 cm of the poles and were generally not maintained above background concentrations over 12 months after treatment (section 2.3.6.2.).

Therefore, where comparisons between field and model were possible, in terms of preservative loadings in the timber, and the degree and extent of preservative soil contamination around preservative treated timber, findings from the model and the field compared favourably. This indicated that the fluoride and chromium concentrations in the drainage waters and the foliar fluoride and chromium concentrations in plants adjacent to the treated pole section in model unit TS, and by implication TSS, were comparable with those found adjacent to treated timber in the field. However, as no such field measurements were carried out, the measurements of these parameters from the uncontaminated model

unit CS were compared with known background field values, to determine whether these model and field values were similar.

The mean concentrations of fluoride in the upper and lower leachates from the CS soilbed, at 0.940 and 0.615 ug/cm^3 respectively (table 3.3.2.5), accorded well with expected background levels of up to 0.95 ug/cm^3 in the field (Larsen and Widdowson, 1971; Russell, 1980). Similarly, the mean concentrations of total chromium in these leachates, at 0.014 and 0.016 ug/cm^3 respectively (table 3.3.2.9), were well within the normal range of up to 0.05 ug/cm^3 found in natural soil drainage waters from uncontaminated field sites (Anon, 1976; Calder, 1988). This was not unexpected, as, in the absence of remedially treated timber in this model unit, these values were dependant on the normal soil concentrations of these elements, and the range of fluoride and chromium soil concentrations found in CS, at 295.44 - 390.40 ug/g and 58.52 - 79.20 ug/g respectively (tables 3.3.5.1 and 3.3.5.2), were comparable with the background mean concentration ranges of these elements, of 234.16 - 327.34 ug/g and 47.80 - 68.85 ug/g respectively (table 2.3.6.5) recorded over a year at the Glenclova field site. As the uncontaminated field model therefore provided soil water concentrations of these elements consistent with those in the field, the increased concentrations of fluoride and chromium found in the drainage waters of model units TSS and TS (section 3.4.2.3.), were likely to be consistent with those adjacent to treated field poles.

Similarly, the foliar fluoride concentrations found in ryegrass samples recovered from the uncontaminated soil adjacent to the pole section in model unit CS, which ranged from 3.84 - 13.87 ug/g (table 3.3.3.1), were well within a range of 4.7 - 209 ug/g for grasses from uncontaminated field sites (Levaggi et al, 1971; McQuaker and Gurney, 1977; Villa, 1979). However, the foliar chromium concentrations in these ryegrass samples, which ranged from 4.83 - 15.80 ug/g (table 3.3.3.1), were in excess of a range of values for vegetation growing on uncontaminated sites (Anon, 1981). This indicated that while the elevated foliar fluoride concentrations recorded for ryegrass adjacent to the treated pole

sections in TSS and TS, were probably comparable with field values, the foliar chromium concentrations in these grass samples were not. However, foliar fluoride and chromium concentrations were not the prime cause of dry matter yield reductions adjacent to remedially treated timber in TSS and TS (section 3.4.3.1.2.). Therefore, the possibility that foliar chromium concentrations adjacent to remedially treated timber in TSS and TS were also possibly higher than expected merely served to highlight how remote was the possibility that these foliar accumulations would find a way into terrestrial food chains.

Therefore, with regard to a number of important environmental values, comparisons between the physical model and the field indicated substantial agreement. Hence, it was reasonable to conclude that the limited environmental effects due to the presence of treated timber in model units TSS and TS, indicated that the environmental impact of remedially treated timber in the field would be slight.

CHAPTER 4.

GENERAL CONCLUSIONS.

4.1. The efficacy and environmental impact of Rentex remedial treatment for creosoted distribution poles.

This work has shown by a combination of field and laboratory studies that Rentex remedial treatment was successful in rapidly establishing fungitoxic concentrations of preservative fluoride within the uncreosoted groundline region of creosoted distribution poles and pole sections, sufficient to prevent or eradicate the growth of the basidiomycete ***Neolentinus lepideus***, the primary causal agent of internal decay in the susceptible groundline region of these timbers (section 2.4.2.4.). Toxic concentrations of diffused fluoride were maintained within the timber for a period of at least 18 months after treatment (section 2.4.2.4.), and possibly longer (section 2.4.3.3.). These findings clearly indicated the efficacy of the remedial treatment over a relatively short timescale.

However, it is unlikely that toxic fluoride concentrations would be maintained over much longer periods, as the chromium component of the preservative, intended to prevent the leaching of diffused fluoride, possibly by the formation of insoluble fluo-chrome complexes, did not diffuse into the timber cross section from the preservative injection sites (section 2.4.3.3.) due to insufficient timber moisture contents and fixation within the timber (section 2.4.3.1.). Hence, diffused fluoride remained highly mobile within the timber, resulting in a decrease in fluoride concentrations within the uncreosoted groundline area at 20 months after treatment due to leaching (section 2.4.3.3.).

Therefore, while the remedial treatment provided a direct preservative effect in creosoted poles and pole sections within the first 2 years after treatment, this effect would decline thereafter. Long term prevention of internal decay would depend on the speed with which the groundline region of treated poles was re-colonised by ***N. lepideus*** and/or other wood decay basidiomycetes. Given that ***N. lepideus***, the basidiomycete most commonly found in field poles in this study, was isolated from approximately 5 % of wood cores recovered from 160 non-remedially treated creosoted field poles which had been in service

for 31 years (tables 2.3.3.2. and 2.3.3.3.), this re-colonisation may occur slowly. Hence, remedially treated distribution poles may remain free of *N. lentinus* for a number of years after fluoride concentrations within the groundline zone are below the toxic threshold to this basidiomycete.

These findings indicated that Rentex remedial treatment would prevent or retard internal decay at the groundline of creosoted electricity distribution poles for a number of years after treatment. Therefore the treatment will actively prolong the service life of distribution poles in service and postpone replacement.

However, the leaching of both fluoride and chromium from remedially treated timber in the field always resulted in significant contamination of adjacent soil (section 2.4.3.2.). The environmental impact of these preservative losses was examined using a representative physical field model incorporating treated timber and a range of physical and biological systems (section 3.1.8.), identified as probable indicators of any detrimental environmental effects of the treatment, by virtue of the normal field exposure of treated distribution poles. Though increased concentrations of preservative fluoride and chromium in soil and drainage waters adjacent to treated timber (significantly reduced ryegrass yields and soil microbial activity, these biological effects extended no further than 10 cm from the timber (section 3.4.4.). Hence, while definite deleterious environmental effects are likely to be found adjacent to remedially treated field poles, their very localised nature suggests that remedially treated distribution poles should not be regarded as a serious environmental hazard.

4.2. Appraisal of the physical model for use in other wood preservative testing protocols.

The model system has been successfully used in this study to allow the simultaneous measurement and linkage of several physical and biological indicators of the environmental impact of remedially treated pole sections (sections 3.4.2. and 3.4.3. respectively). Moreover, the similarity between conditions maintained in the physical models and those in the field, indicated that the effects of treated timber in the model unit environments and in the field would be substantially the same (section 3.4.4.). Therefore, the physical model represents a useful tool for detailed laboratory based environmental studies of wood preservatives.

This model has many advantages over traditional environmental field studies, including the speed with which experimentation can be carried out and the degree of control possible over experimental conditions. For instance, the seeding and accurate sampling of plant material around treated structures in the field would require that the chosen site be protected from grazing by domestic and other animals (see section 3.1.7.1.), and even if these activities were successfully excluded from the site, seasonal constraints are imposed on plant studies in the field.

Similarly, while field studies are time consuming and expensive, the model represents a much more convenient, accessible and economic option. These features of the model were probably best demonstrated by the ease with which contaminated drainage water was collected for analysis. The acquisition of similar samples in the field would involve the installation of a drainage system adjacent to the treated structure with considerable time expended in excavation work (see section 3.1.7.1.).

Though lysimeters, which consist of undisturbed soil monoliths, also possess many of the same advantages over field studies and have been widely used in eco-toxicological

studies, they are very expensive (£2,000) (section 3.1.7.2.) by comparison with the model designed for this project (< £300). In addition, the undisturbed nature of lysimeters would not allow the installation of the complicated drainage system necessary to examine contaminants in the waters close to the important rhizosphere area near the soil surface.

4.3. Further development of the physical model for inclusion in preservative testing protocols.

Despite its obvious advantages over field studies (section 4.2.), this model design was essentially a prototype. Though the studies which were carried out to assess environmental effects adjacent to remedially treated timber were successful, the drainage system did not function as well as expected, and certain aspects of plant growth were adversely affected by the model conditions.

The drain design was based on standard agricultural field drains (section 3.2.2.2.) and each was constructed of flexible perforated PVC piping topped with a layer of permeable gravel to facilitate water entry. These drains operate on the principle that when the surrounding soil is saturated with water, the water follows the course of least resistance and enters the pipe. However, the topsoil drains 1, 2, 3 and 4 (figure 3.2.2.) did not flow for any model unit, due to improved soil drainageto lower drains via gravel amendment, and the absence of these leachates limited measurements of the fluoride and chromium content in the important rhizosphere area. In addition, several operative drains were subject to soil blockage which prevented leachate collection on some occassions and encouraged variability in leachate volume (section 3.3.1.3.1.).

To make leachate collection more reliable and remove the requirement for soil saturation to acquire samples, it is proposed that the simulated field drain design be largely discarded. A simpler design consisting of a system of rigid open **V** or **U** shaped gutters,

in-filled with gravel to prevent soil blockage and similarly positioned within the soilbed, would make the drains independent of soil saturation and less prone to soil blockage. Hence, leachates could be collected from the rhizosphere area of the soilbed irrespective of the soil type used and sample volumes from individual drains would probably remain more stable between collections. This drain design would certainly improve leachate collection in the upper part of the soilbed.

However, with regard to the base drains 5 and 9 (figure 3.2.2.), these would always be in an area of saturation and the present flexible and transparent drain material was useful in allowing the water table in each model unit to be altered and its height to be visually checked. Therefore, it is proposed that the perforated portion of these drains within the model unit, which were subject to periodic blockage, be discarded, and that the drain ports be moved from the side of each model unit (figure 3.2.2.) to the base. Hence, these drains would simply consist of holes in the model base, protected by a gravel in-fill, emptying into flexible ports as before. This would improve drainage flow and retain control over watertable height.

With respect to plant growth within each soilbed, the chlorosis observed in the young ryegrass swards and their premature death after cutting (section 3.2.4.4.) was symptomatic of plants grown in waterlogged soils (Russell, 1977). Similarly excess water was implicated in the poor root development of rye plants (section 3.4.3.2.2.). Evidently, the rainfall simulations, raised watertable and hand watering operations necessary to maintain the soil within each soilbed at field capacity to encourage leaching of preservative from the treated timber, also provided soil conditions which were stressful for plant growth.

To lessen this apparent incompatibility, the seeding and establishment of crops within the soilbeds could be carried out in drier soil conditions more conducive to plant growth, before positioning the treated timber. This would encourage the development of a more deeply rooted crop prior to the imposition of moist soil conditions. The improved soil cover

provided by the larger crop would lessen evaporation from the soil surface and less water input would be required. While these plants would still be exposed to excessive soil moisture contents, their deeper rooting habit would provide a more open aerated soilbed, allowing the plants to better withstand the conditions.

The modifications to the design and operation of the physical field model brought about by the re-design of the drainage system and earlier plant establishment will promote greater reproducibility of results and go some way towards standardising the design for its further use in efficacy and environmental tests of other wood preservatives.

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APPENDIX 1

Efficiency of the modified alkali fusion technique for extraction of fluoride and chromium from samples of wood and soil.

Summary.

Known fluoride and chromium additions were made to wood and soil samples. The samples were fused according to the modified extraction method (section 2.2.2.3.3.) and the aqueous extracts produced were analysed for fluoride and chromium content (sections 2.2.2.3.4. and 2.2.2.3.5.). The concentrations of each element found in the spiked samples indicated that the modified technique was an efficient method for extraction of fluoride and chromium from wood and soil.

Apparatus.

Total chromium measurements were carried out using a Perkin Elmer 1100B atomic absorption spectrophotometer, and fluoride measurements were carried out using a Corning Eel model 12 pH meter equipped with a Russell model 94-4099 fluoride electrode and reference electrode type 900019.

Reagents.

All analytical solutions were as used for wood samples (section 2.2.2.3.2.) and were prepared using chemicals of Analar quality and grade A glassware. A standard fluoride solution (1000 ugF/cm^3) for standard additions consisted of sodium fluoride (2.2100 g) dissolved and made up to 1 dm^3 in distilled water. For smaller additions, another fluoride solution (100 ugF/cm^3) was prepared by dilution of the standard. A standard chromium

solution consisted of sodium dichromate (2.8660 g) dissolved and made up to 1 dm^3 in distilled water. Other chromium solutions (100 and 25 ugCr/cm^3) were prepared by dilution of the standard.

Procedure.

Samples for fluoride and chromium additions consisted of milled Scots pine heartwood and a sandy loam soil finely ground in a mortar and pestle and sieved through a 2 mm mesh stainless steel sieve. Each sample type was thoroughly homogenised and 0.25 g amounts of each were weighed into separate nickel crucibles (70 cm^3).

Combined additions of fluoride and chromium were pipetted into crucibles as indicated (tables 1 and 2). No additions were made to a number of wood and soil crucible samples. All samples were dried overnight in an oven set at 30°C. Fluoride and chromium were extracted from the samples into solution using the modified alkali fusion method (section 2.2.2.3.3.) and measured using the method of standard additions by ion selective electrode (section 2.2.2.3.4.) and atomic absorption spectrophotometry (section 2.2.2.3.5.) respectively. Solutions from samples to which no additions were made were combined and used for background correction as reagent blanks.

Results.

Mean recoveries of fluoride and chromium additions from each sample type are presented in tables 1 and 2, with standard deviations for means of 5 in parenthesis. Mean percentage recoveries are also presented. The results clearly indicate the efficiency of the modified technique for extraction of fluoride and chromium (section 2.2.2.3.3.) from samples of wood and soil.

Table 1. Mean percentage recoveries of fluoride added to samples of wood and soil (standard deviations in parenthesis are for means of 5).

Sample Type	Sample Number	Fluoride Added (ug)	Mean Recovery of added Fluoride (ug)	Fluoride Recovery (%)	Mean Recovery (%)
Scots Pine Heartwood	**1**	200	0192.52 (06.54)	96.26	**93.24**
	2	400	0367.58 (14.24)	91.89	
	3	1000	0915.72 (53.87)	91.57	
Sandy Loam Soil	**1**	100	0091.75 (02.30)	91.75	**90.81**
	2	500	0453.48 (08.42)	90.70	
	3	1000	0896.71 (25.19)	89.67	
	4	2000	1822.27 (40.88)	91.11	

Table 2. Mean percentage recoveries of chromium added to samples of wood and soil (standard deviations in parenthesis are for means of 5).

Sample Type	Sample Number	Chromium Added (ug)	Mean Recovery of added Chromium (ug)	Chromium Recovery (%)	Mean Recovery (%)
Scots Pine Heartwood	**1**	50	0047.92 (01.22)	95.84	**92.32**
	2	250	0216.80 (04.00)	86.72	
	3	1000	0944.10 (29.31)	94.41	
Sandy Loam Soil	**1**	100	0091.20 (04.27)	91.20	**92.48**
	2	500	0450.72 (09.01)	90.14	
	3	1000	0913.82 (29.47)	91.40	
	4	2000	1943.81 (37.54)	97.19	

APPENDIX 2.

Experimental check of the accuracy of the selective ion electrode method for determination of fluoride concentration in leachate samples.

The efficiency of the selective ion electrode method (section 2.2.2.3.4.) for determining the fluoride content of leachate samples of variable pH (section 3.2.5.1.3.) was tested as follows:

Portions from a number of individual leachate samples (section 3.2.4.3.) were mixed to provide a solution of pH 5.75 and another of pH 7.05, measured with a Corning Eel Model 12 pH meter and electrode (section 3.2.5.1.2.). Representative samples (20 cm^3) of each solution were pipetted into 8 polyethylene beakers of 100 cm^3. One, 2 and 3 cm^3 portions of a fluoride solution (50 ugF/cm^3), made up by dilution of a standard fluoride solution (appendix 1), were pipetted into each of 2 of the 8 sample solutions. All were made up to 25 cm^3 with pipetted amounts of distilled water. This provided 8 sample solutions of 25 cm^3 for each mixed leachate solution, 2 containing leachate only and 2 of each of the remaining 6 containing 50, 100 and 150 ug of added fluoride. Twenty five cm^3 of TISAB buffer (section 2.2.2.3.2.) was added to each and measurement of the fluoride content of each solution carried out (section 2.2.2.3.4.), using the samples which had received no addition to determine a mean blank value.

Calculated mean percentage recoveries and standard deviations of added 50, 100 and 150 ug of fluoride, equivalent to 1, 2 and 3 ugF/cm^3, were 103.89 (0.81), 104.58 (0.94) and 100.05 (0.00) respectively for leachate samples of pH 5.75 and 100.89 (5.64), 101.82 (1.07) and 99.80 (1.79) respectively for leachate samples of pH 7.05. These findings indicated that, for leachate samples (section 3.2.4.3.), accurate measurements of low fluoride concentrations could be carried out, over a range of pH, using the selective ion

electrode method (section 2.2.2.3.4.).

APPENDIX 3.

Experimental check of the accuracy of the atomic absorption addition calibration method for determination of the total chromium concentration of leachate samples.

Introduction.

It is recommended (*Anon, 1982*) that for measurement of the chromium concentration in aqueous samples, using atomic absorption spectrophotometry, the element should be present exclusively in only one oxidation state. Hence, the modified alkali fusion method for extraction into solution of fluoride and chromium in wood and soil samples (section 2.2.2.3.3.) includes the addition of sodium peroxide for oxidation of all chromium present to the chromium (VI) form before measurement (section 2.2.2.3.5.). However, leachate samples (section 3.2.4.3.) would typically contain chromium (VI), the form of the element present in Rentex, and chromium (III), its reduced form in the presence of soil organic matter. It was therefore necessary to determine whether accurate measurement of the total chromium content was possible when both oxidation states of the element were present in aqueous samples. In addition, measurements of the chromium contents of leachate and 'rainfall' samples were carried out using samples which were not acidified and with no reagent blank (section 3.2.5.1.4.). Thus, it was also necessary to determine whether variations in sample pH and the use of distilled water in place of a reagent blank would have a negative effect on the accuracy of chromium measurements.

Apparatus.

Perkin Elmer 1100B atomic absorption spectrophotometer as used for measurement of total chromium contents of wood sample extraction solutions (section 2.2.2.3.5.).

Reagents.

Analar reagents and grade A glassware were used for preparation of all analytical solutions. Standard solutions of chromium (VI) and chromium (III), both 1000 ugCr/cm^3, were prepared by dissolving sodium dichromate (2.8660 g) and chromic chloride (5.1240 g) respectively, in separate 1 dm^3 volumetric flasks containing distilled water (300 cm^3) and made up to 1 dm^3. A solution for acidification of samples was prepared as for wood sample extraction solutions (section 2.2.2.3.2.).

Method.

A set of volumetric flasks (each 25 cm^3) were numbered 1-9 and acidifying solution (1 cm^3) was added to flasks 5, 7 and 9. All flasks were made up to volume with distilled water. Chromium additions were made using a BCL solid displacement micropipette as follows:

Flask 1	No addition	-	blank solution of distilled water
" 2	25 ul of Cr(VI) solution		=> 1 ugCr(VI)/cm^3
" 3	50 " " " "		=> 2 "
" 4	75 " " " "		=> 3 "
" 5	25 " " " "		=> 1 "
" 6	25 " " Cr(III) "		=> 1 ugCr(III)/cm^3
" 7	25 " " " "		=> 1 "
" 8	25 " each of Cr(VI)/(III) solutions		=> 2 ugCr(VI)/(III)cm^3
" 9	25 " " " " "		=> 2 "

Three replicates of each of the 9 flasks were prepared. Flask 2 represented the initial sample for measurement and flasks 3 and 4 represented standard additions to this sample. A further set of 3 replicates was set up using a leachate solution in place of distilled water in flasks 2 - 9. This solution was prepared by combining portions of a number of control leachate samples (section 3.2.4.3.). Chromium measurement of flask solutions 2, 5, 6, 7, 8 and 9 from each group of 9 flasks was carried out according to the atomic absorption addition calibration method (section 2.2.2.3.5.).

Results.

Mean percentage recoveries of added chromium, with standard deviations in parenthesis, for flask solutions made up in distilled water, A, and for flask solutions made up using a control leachate, B, were as follows:

Flask	A	B
2	95.3 (3.47)	93.4 (1.65)
5	95.7 (2.26)	97.4 (1.85)
6	96.7 (1.66)	96.4 (1.35)
7	94.8 (1.18)	98.2 (2.25)
8	92.8 (1.86)	96.4 (2.95)
9	93.3 (2.15)	96.8 (1.00)

These results indicated that accurate measurement of the total chromium content in aqueous samples by the atomic absorption addition calibration method (section 2.2.2.3.5.) was not significantly affected by the oxidation states of chromium in the samples or the pH of samples. In addition the use of a distilled water blank solution in place of a reagent blank did not adversely affect chromium recovery from leachate samples. Thus the total chromium content of filtered soil leachate and 'rainfall' samples (section 3.2.5.1.1.) could be accurately measured, without further preparation, using a blank solution of distilled water (section 3.2.5.1.4.).

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Sinclair, D. C. R., Smith, G. M., Bruce, A., King, B. and Staines, H. J. (1991). Diffusion of Chromium and Fluoride in Rentex Treated Creosoted Pole Sections. The Inter. Res. Group on Wood Preserv. Document No.: **IRG/WP/3659.**

Sinclair, D. C. R., Smith, G. M., Bruce, A. and King, B. (1992). Development of a Model System to Assess the Efficacy and Environmental Impact of a Chromated Fluoride Remedial Treatment for Creosoted Distribution Poles. The Inter. Res. Group on Wood Preserv. Document No.: **IRG/WP/2395-92.**

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Diffusion of chromium and fluoride in Rentex treated creosoted pole sections.

by

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DIFFUSION OF CHROMIUM AND FLUORIDE IN RENTEX TREATED CREOSOTED POLE SECTIONS.

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ABSTRACT.

A chromated fluoride preservative was applied by injection to the groundline regions of creosoted distribution pole sections and these were erected at a filed site in Scotland. Wood samples were recovered for chemical analysis at one week, two, five and twelve months after treatment. Small sample mass and destructive nature of the analysis necessitated modification of an alkali fusion technique to allow a single analysis for both fluroide and chromium. This paper details the methodology employed and reports on the extent to which diffusion of chromium and fluoride has occurred. The implications on the efficacy of the preservative formulation for use in distribution poles is discussed.

Keywords: Creosoted poles; remedial treatments; Cobra process; fluoride; chromium.

INTRODUCTION.

The preservative Rentex contains as its main toxic constituents sodium fluoride, ammonium bifluoride and sodium dichromate. It has been investigated (Bruce and King 1989) as a successor to Cobra (DFA) salts, a formulation containing dinitrophenol, sodium fluoride and arsenic (111) oxide used until recently as a remedial groundline treatment to control internal decay in creosoted distribution poles.

Rentex is a waterborne preservative which can be produced as a paste for use by the Cobra injection process. Application is carried out by forcing the paste into the pole to a depth of 65mm via a hollow injection needle propelled by a mechanical pump. Pole diameter determines the number of injections (up to 120). applied in a defined pattern to the pole in a treatment zone which extends approximately 35cm above and 35 cm below the groundline. A bitumen coating is applied to the whole treated region and an aluminium sheath is placed round the treated pole section above the groundline.

The use of fluorides as waterborne preservatives in conjunction with chromium compounds is well documented (Steinherz 1939, Becker 1970). Efficacy is dependant on the penetrability and permanence of the applied fluoride. These parameters have been examined (Smith and Cockcroft 1967) using distribution poles treated with Cobra salts.

Results of the chemical analyses of cores taken from a population of imperfectly creosoted Baltic redwood poles removed 2 years after Cobra (DFA) injection indicated fluoride diffusion had occurred throughout the sapwood of the treated zone (Smith and Cockcroft 1967). Loadings of fluoride above the toxic values for Lentinus lepideus, the major internal decay fungus, were established. When sampled 4 years after treatment a systematic loss of fluoride to below toxic values was observed. A corroborative study (Henningson and Nilsson 1975) has indicated fluoride loadings giving sterilisation of the treatment zone within 1 year of application and progressive loss of protection thereafter till at 9 years both treated and control poles contained the same microflora.

A recent field evaluation of Rentex (Bruce and King 1989) showed no re-isolation of L.lepideus within 15 months of treatment in creosoted pole sections artificially inoculated with the decay fungus prior to treatment.

The studies of Smith and Cockcroft (1967) and Henningson and Nilsson (1975) to determine the efficacy of Cobra (DFA) salts employed a colorimetric analysis to estimate fluoride content of wood based on the method of Megregian (1954). The fluoride selective ion electrode is now the preferred tool for analysing fluoride, used in conjunction with different extraction methods. However the determination of fluoride and chromium has not recently been attempted and methodology is sparse. Accordingly an alkali-fusion extraction technique for fluoride analysis of soils and vegetation (Mcquaker and Gurney 1977) was modified for the current work to include extraction into solution of the chromium in the samples. Fluoride concentrations were then determined by ion selective electrode and chromium by atomic absorption spectrophotometry.

A study of the efficacy and environmental impact of the use of Rentex by the Cobra process as a remedial preservative for distribution poles has been set up at Dundee. This paper describes a part of this work in which a field experiment is used to study the distribution of fluoride and chromium in creosoted distribution poles over a period of time after Rentex has been applied as a groundline remedial treatment for control of internal decay.

MATERIALS AND METHODS.

Eighteen 6m pole lengths of medium diameter were cut from the mid to upper sections of aged creosoted service poles. The top end of each length was tapered and coated with bitumen to facilitate rainwater run-off. The pole lengths were erected at a field site near Dundee and treated with Rentex paste at the groundline region by the Cobra process. All treatment zones were bitumen coated and sheathed in aluminium above the groundline. The Rentex Paste contained sodium dichromate, ammonium bifluoride, sodium fluoride and some compounds to adjust pH, ionic strength, and paste consistency.

1. Sampling.

Two of the 18 erected pole lengths were randomly selected, removed immediately and stored under cover for 1 week prior to sampling. The treated zones were cut from each of these two pole lengths (approximately 175mm above and below the groundline position). Moisture contents were recorded at the exposed surfaces using a Protimeter moisture meter. Two 1 cm deep discs were cut midway between the groundline level and the top and bottom surfaces of the treated zone (Fig. 1). Wood samples were then obtained from the uncreosoted regions of the discs by carefully splitting along the injection lines as shown in Figure 1. The four wood samples thus produced from each disc for each position labelled 1-8 were then ground up and bulked together to give a representative sample for each position 1 to 8. Duplicate analyses were then carried out for each sample. A further 2 poles were recovered for sampling and analysis at 2, 5 and 12 months time intervals after treatment.

2. Analysis.

Fluoride measurements were carried out using a Corning eel model 12 pH meter equipped with a Russell model 94-4099 fluoride electrode and reference electrode type 900019. Chromium measurements were made using a Perkin Elmer 1100B atomic absorption spectrophotometer.

Total ionic strength adjustment buffer (TISAB) consisted of 58 ml of glacial acetic acid and 12g of sodium citrate dissolved in 300 ml of distilled water adjusted to pH 5.2 using 5M NaOH, made up to 1 litre. Acidifying solution contained 200ml of 2.5M sulphuric acid and 100ml of 30g/litre sodium sulphate solution made up to 1 litre. All reagents were of Analar quality.

Approximately 0.25g of sample (corrected for moisture content) was weighed into a nickel crucible (70 ml) and 10ml of 5M NaOH added. The crucible was heated for 40 minutes on a hotplate set at 150°C. The dried sample was placed over a Meker (Amal major) burner and brought to red heat over a 5 minute period using a low flame initially to avoid combustion. Red heat was maintained for 30 seconds on full flame, and the crucible removed to allow contents to solidify. The solidified sample was broken up by gentle heating after addition of 10ml of distilled water. One gram of sodium peroxide was quickly added with vigorous stirring. The mixture was allowed to cool and 6ml of concentrated HCl added slowly with stirring to adjust the pH to 8-9(monitored with the use of pH paper). The contents of the crucible were left to cool then washed into a 100ml volumetric flask, through Whatman no. 4 filter paper, and made up to the volume with distilled water.

Fluoride measurement was carried out by adding 25ml of the sample solution to 25ml of TISAB buffer and recording the electrode potential using the fluoride and reference electrodes. A further two potentials were recorded after adding known amounts of standard fluoride solution and the concentration of fluoride in the original 100ml sample was determined by the double known addition method.

Chromium measurement was carried out by atomic absorption spectrophotometry. A portion of the sample solution was diluted to fall within the working concentration range for chromium (up to 5ppm) and acidified with sodium sulphate/sulphuric acid solution. The concentration of chromium in the original sample was then obtained using the method of standard additions.

RESULTS.

The mean % w/w concentration and standard deviations of the chromium and fluoride in the wood samples are shown in tables 1 and 2. The results for individual poles at the two disc heights at each time period were combined and represented in 3 dimensions as shown in the diagram of Figure 2. Three diagrams each for the chromium and fluoride concentrations at each sample position were produced; A) 175mm below and B) 175mm above the groundline and a mean of these C) as shown in figures 3, 4, 5 and 6.

In addition the concentration values of the elements at each of the 8 sample positions in these figures were combined and a mean taken to produce Figure 7. In order to estimate the movement of the fluoride away from the sites at the injection lines ie. sites 1 and 4 the fluoride concentrations at all the sample positions (excluding 1 and 4) of each fluoride model in Figures 3-6 were combined and the mean expressed as a percentage of each fluoride mean in Figure 7 (Figure 8 ABC). This percentage is compared with the mean of all sample positions excluding 1 and 4 in figure 8DEF.

A comparison of the mean % element concentrations in figure 3 clearly shows that the fluoride and chromium levels above ground are greater than those at the below ground sample position at zero time i.e. one week after injection time. This is attributed to the method of injection used to apply the preservative paste. This occurs because sampling above the groundline permits greater leverage thereby enabling greater pressure to be applied by the mechanical pump.

A comparison of figures 3 to 6 for the chromium levels shows a marked decrease for the above groundline samples especially over the first two month period. There is a more gradual chromium decrease with time below the groundline. These comparisons are also seen in figure 7.

Figures 3 to 6 also show that there is little movement of chromium in the tangential direction. The apparent increase in chromium content in the radial direction at the 5 month interval (Fig. 5) is attributed to the smaller diameter of pole 6 (16.2 cm) sampled at this time.

Figures 3 to 6 show that fluorine is moving outwards from the injection site in the transverse plane of the pole over the 12 month exposure period. Indeed its movement is evident between the time of actual injection and the 'zero time' pole sampling one week later (Fig.3).

Figure 7 shows that there is a general decrease in fluoride content over the 12 month period for the above groundline sample positions. The decrease is greatest over the first 5 month period of field exposure. At the below ground sampling position there is an interesting increase in mean fluoride content (fig. 7) over the first two months which then returns to its original value after 5 months and remains there for at least the next 7 months.

The mean % Fluoride concentrations at sample positions away from the injection lines (D E F fig. 8) increase from zero to 5 months and then level out at about 0.08 % fluoride. The proportion of total fluoride in these areas (A B C fig.8) also increases up to 5 months and then levels out at about 35% of the total. While this data indicates definite fluoride movement into the adjacent transverse plane, A B and C are percentage values of the total fluorine which decreases over this period.

DISCUSSION.

The most outstanding feature of the analytical data represented in figure 3 is the contrast between the concentrations of both fluoride and chromium above and below the groundline. The difference is almost certainly due to the restricted use of the pump mechanism which squeezes the preservative paste through the hollow injection needle for the below ground samples. Over the 12 month sampling period these differences are gradually evened out for both elements as shown in figs. 3 to 6.

A considerable decrease in mean % chromium concentration particularly over the first two months for the above ground sampling position is clear from fig.7. A similar trend is also seen for the fluoride concentrations above the groundline. This is in line with earlier work (Henningson and Nilsson, 1975) who reported migration of fluoride both to the interior of the pole and to the surrounding soil from poles treated with Cobra (DFA) salts.

The lack of movement of chromium into sample positions away from the injection sites coupled with the considerable decrease in its mean concentration above the groundline over the first two months indicates that a proportion of this element is lost from the poles by a leaching process. The remaining proportion is probably fixed in the wood near the injection site. It is also clear that the chromium cannot "fix" the fluoride in those regions of the pole away from the injection sites which it does not reach. The fixation properties of chromium (V1) for some elements e.g. arsenic are well known particularly in connection with the use of CCA but not so well established for fluoride (Becker,1973, Nicholas 1971). Nicholas (1971) suggests that there is a limited number of active sites available in wood for the adsorption of chromium ions and that these are quickly saturated. Once saturation has occurred the excess chromium could be leached. In view of the limited region of wood available to the chromium for adsorption along the injection line and the lack of diffusion of this element in directions away from the injection site it is likely that leaching has taken place along the injection path.

Figures 3-6 and also fig. 8 clearly show the mobility of fluoride in the wood. While the above groundline sampling position showed a decrease in mean % fluoride concentration, the below groundline sampling position showed no decrease over the 12 month period and actually increased during the first two months. These findings suggest that there is a net downwards movement of fluoride in the poles and some loss of this element to the surrounding soil.

As well as preservative permanence, the efficacy of pole remedial treatment by injection with Rentex will be dependant on diffused fluoride reaching toxic levels for the decay fungi at the interior regions of the poles. Henningson and Nilsson (1975) quoted a value of $1Kg/m^3$ (0.2% w/w) for fluoride from Liese and Groger (1954) as representing a threshold value for protection of wood against decay fungi. Smith and Cockcroft (1967) however, quote a number of papers which give differing ranges (0.005 lb/ft^3- 0.02lb/ft^3corresponding to 0.016-0.064% w/w) for the toxic values of sodium fluoride against Lentinus lepideus.

In conclusion the results of this study show that fluoride has migrated from injection positions throughout the groundline treatment zone. Work is currently in progress to establish whether toxic levels of Rentex against a range of decay fungi are reached in the interior regions away from the injection sites. The fact that the results of this study also suggest that both fluoride and chromium are being leached into the surrounding soil give cause for concern, both in terms of the long term permanence and protection afforded by the preservative and also with regard to the environmental consequences of this loss.

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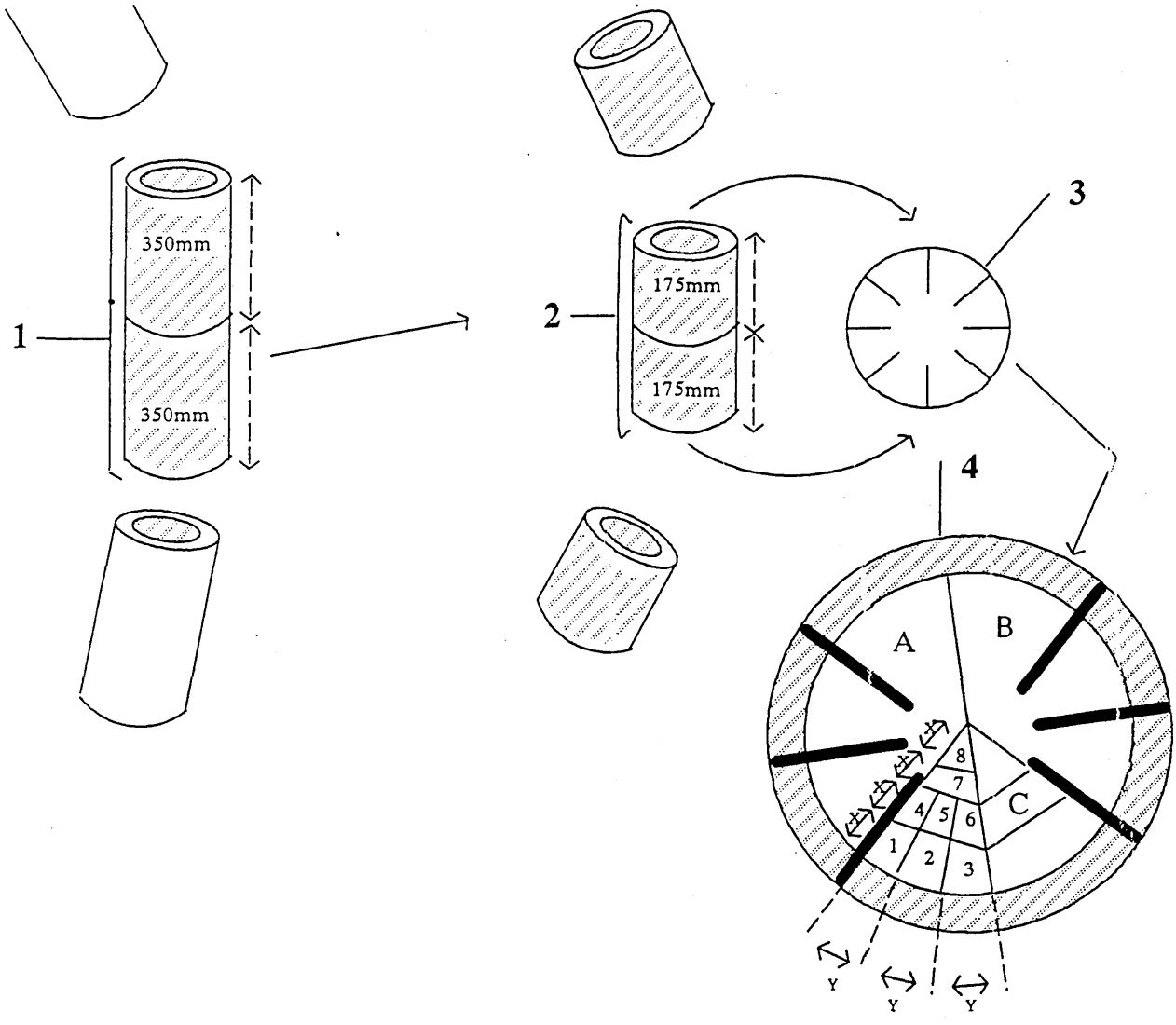

Figure 1. Procedure for removal of wood samples 1 - 8 from the uncreosoted area of the remedially treated zone of distribution pole sections for fluoride and chromium distribution studies (see section 2.2.2.2.2. for description).

POLE NO.1 | 21.2cm dia.| MOISTURE. 175mm+/175mm- = 21/21% | 0 MONTHS.							
175mm above ground level.				175mm below ground level.			
	MEAN %F	(SD)	MEAN %Cr	(SD)	MEAN %F	(SD)	MEAN %Cr	(SD)
1	2.9086	(0.0180)	2.3760	(0.0218)	1.4155	(0.0687)	1.2440	(0.1189)
2	0.1041	(0.0094)	0.0513	(0.0038)	0.0286	(0.0000)	0.0001	(0.0001)
3	0.1244	(0.0013)	0.0669	(0.0102)	0.0295	(0.0021)	0.0004	(0.0001)
4	2.0530	(0.0038)	1.9660	(0.1620)	0.5596	(0.0011)	0.5403	(0.0020)
5	0.0430	(0.0924)	0.0072	(0.0067)	0.0289	(0.0022)	0.0000	(0.0000)
6	0.0285	(0.0001)	0.0021	(0.0002)	0.0297	(0.0004)	0.0024	(0.0034)
7	0.0391	(0.0001)	0.0032	(0.0006)	0.0272	(0.0001)	0.0006	(0.0008)
8	0.0552	(0.0000)	0.0060	(0.0037)	0.0272	(0.0004)	0.0000	(0.0000)
POLE NO.2 | 18.8cm dia.| MOISTURE. 175mm+/175mm- = 14/14% | 0 MONTHS.							
1	1.8350	(0.1560)	1.8289	(0.0799)	0.5282	(0.0192)	0.2790	(0.0622)
2	0.0456	(0.0108)	0.0042	(0.0009)	0.0888	(0.0000)	0.0104	(0.0000)
3	0.0388	(0.0002)	0.0019	(0.0009)	0.0338	(0.0002)	0.0038	(0.0004)
4	0.4677	(0.0004)	0.2498	(0.0253)	0.1817	(0.0002)	0.0478	(0.0009)
5	0.0294	(0.0002)	0.0042	(0.0009)	0.0474	(0.0042)	0.0004	(0.0006)
6	0.0302	(0.0007)	0.0045	(0.0014)	0.0741	(0.0000)	0.0287	(0.0000)
7	0.0331	(0.0004)	0.0013	(0.0018)	0.0329	(0.0004)	0.0005	(0.0008)
8	0.0251	(0.0000)	0.0038	(0.0000)	0.0227	(0.0000)	0.0001	(0.0000)
POLE NO.3 | 18.5cm dia.| MOISTURE. 175mm+/175mm- = 16/18% | 2 MONTHS.							
1	1.6035	(0.1286)	0.3924	(0.0302)	1.3209	(0.1316)	0.5430	(0.0091)
2	0.0444	(0.0001)	0.0000	(0.0000)	0.0641	(0.0000)	0.0008	(0.0003)
3	0.0324	(0.0001)	0.0000	(0.0000)	0.0475	(0.0002)	0.0001	(0.0000)
4	1.7022	(0.0005)	0.4297	(0.0254)	1.4252	(0.0071)	0.5880	(0.0128)
5	0.0295	(0.0000)	0.0000	(0.0000)	0.2273	(0.0006)	0.0049	(0.0022)
6	0.0328	(0.0008)	0.0000	(0.0000)	0.0376	(0.0001)	0.0001	(0.0001)
7	0.2075	(0.0009)	0.0331	(0.0019)	0.0637	(0.0014)	0.0012	(0.0004)
8	0.0357	(0.0051)	0.0006	(0.0008)	0.0486	(0.0001)	0.0005	(0.0007)
POLE NO.4 | 19.7cm dia.| MOISTURE. 175mm+/175mm- = 17/17% | 2 MONTHS.							
1	2.2075	(0.0420)	0.9220	(0.1570)	0.9270	(0.0290)	0.4791	(0.0538)
2	0.0623	(0.0001)	0.0001	(0.0001)	0.0417	(0.0027)	0.0000	(0.0000)
3	0.0425	(0.0001)	0.0000	(0.0000)	0.0401	(0.0001)	0.0000	(0.0000)
4	0.7787	(0.0009)	0.2410	(0.0200)	0.3865	(0.0139)	0.0940	(0.0145)
5	0.1178	(0.0001)	0.0002	(0.0001)	0.0429	(0.0026)	0.0002	(0.0003)
6	0.0445	(0.0001)	0.0004	(0.0001)	0.0310	(0.0006)	0.0002	(0.0003)
7	0.0532	(0.0005)	0.0002	(0.0003)	0.0287	(0.0000)	0.0000	(0.0000)
8	0.0393	(0.0005)	0.0000	(0.0000)	0.0319	(0.0001)	0.0003	(0.0003)

TABLE 1. - Mean percentage Fluorine and Chromium contents of wood from sample positions 1-8, 0 and 2 months after treatment. Standard Deviations (SD) are for means of 2 replicates.

POLE NO.5 | 22.6cm dia.| MOISTURE. 175mm+/175mm- = 22/24% | 5 MONTHS.							
175mm above ground level.				175mm below ground level.			
	MEAN %F	(SD)	MEAN %Cr	(SD)	MEAN %F	(SD)	MEAN %Cr	(SD)
1	0.3576	(0.0366)	0.1688	(0.0163)	0.5636	(0.0016)	0.1424	(0.0323)
2	0.0754	(0.0010)	0.0021	(0.0004)	0.0896	(0.0023)	0.0013	(0.0000)
3	0.0283	(0.0004)	0.0001	(0.0001)	0.0161	(0.0017)	0.0000	(0.0001)
4	0.0652	(0.0007)	0.0014	(0.0002)	0.0659	(0.0006)	0.0008	(0.0002)
5	0.0312	(0.0005)	0.0002	(0.0003)	0.0193	(0.0037)	0.0004	(0.0002)
6	0.0020	(0.0003)	0.0008	(0.0012)	0.0209	(0.0001)	0.0003	(0.0004)
7	0.0128	(0.0001)	0.0007	(0.0008)	0.0149	(0.0001)	0.0002	(0.0002)
8	0.0114	(0.0000)	0.0003	(0.0001)	0.0024	(0.0000)	0.0002	(0.0002)
POLE NO.6 | 16.2cm dia.| MOISTURE. 175mm+/175mm- = 29/30% | 5 MONTHS.							
1	0.9308	(0.0118)	0.3718	(0.0113)	1.0102	(0.0013)	0.5035	(0.0066)
2	0.0490	(0.0017)	0.0005	(0.0001)	0.0631	(0.0007)	0.0008	(0.0002)
3	0.0306	(0.0021)	0.0003	(0.0000)	0.0192	(0.0005)	0.0002	(0.0001)
4	1.5915	(0.0712)	0.5696	(0.0483)	0.7385	(0.0296)	0.5387	(0.0035)
5	0.1606	(0.0000)	0.0023	(0.0000)	0.1047	(0.0032)	0.0032	(0.0004)
6	0.0315	(0.0038)	0.0005	(0.0002)	0.0476	(0.0015)	0.0006	(0.0001)
7	0.4396	(0.0119)	0.3174	(0.0067)	0.2268	(0.0360)	0.0395	(0.0022)
8	0.4448	(0.0000)	0.2805	(0.0000)	0.2253	(0.0166)	0.0367	(0.0016)
POLE NO.7 | 23.6cm dia.| MOISTURE. 175mm+/175mm- =26.5/28% | 12 MONTHS.							
1	0.4745	(0.0657)	0.1841	(0.0082)	0.6818	(0.0773)	0.2864	(0.0375)
2	0.0816	(0.0020)	0.0005	(0.0000)	0.1505	(0.0016)	0.0027	(0.0005)
3	0.0685	(0.0009)	0.0003	(0.0003)	0.1051	(0.0015)	0.0005	(0.0001)
4	0.3088	(0.0037)	0.0238	(0.0062)	0.3698	(0.0035)	0.1460	(0.0210)
5	0.0792	(0.0005)	0.0003	(0.0000)	0.1321	(0.0005)	0.0011	(0.0000)
6	0.0457	(0.0009)	0.0000	(0.0000)	0.0912	(0.0040)	0.0005	(0.0004)
7	0.0330	(0.0024)	0.0001	(0.0002)	0.0444	(0.0016)	0.0004	(0.0006)
8	0.0203	(0.0000)	0.0001	(0.0000)	0.0174	(0.0003)	0.0003	(0.0003)
POLE NO.8 | 26.7cm dia.| MOISTURE. 175mm+/175mm- = 16/18% | 12 MONTHS.							
1	1.2260	(0.2850)	0.7210	(0.1790)	0.8082	(0.1004)	0.2698	(0.0675)
2	0.0926	(0.0040)	0.0012	(0.0006)	0.0648	(0.0035)	0.0004	(0.0003)
3	0.0502	(0.0037)	0.0005	(0.0001)	0.0330	(0.0001)	0.0001	(0.0001)
4	0.6998	(0.0138)	0.3615	(0.0427)	0.3865	(0.0167)	0.2247	(0.0019)
5	0.0526	(0.0018)	0.0000	(0.0000)	0.0843	(0.0041)	0.0006	(0.0000)
6	0.0571	(0.0015)	0.0003	(0.0003)	0.0454	(0.0031)	0.0005	(0.0004)
7	0.2063	(0.0319)	0.0269	(0.0008)	0.0956	(0.0065)	0.0026	(0.0003)
8	0.2418	(0.0962)	0.0369	(0.0198)	0.0209	(0.0004)	0.0000	(0.0000)

TABLE 2. - Mean percentage Fluorine and Chromium contents of wood from sample positions 1-8, 5 and 12 months after treatment. Standard deviations (SD) are for means of 2 replicates.

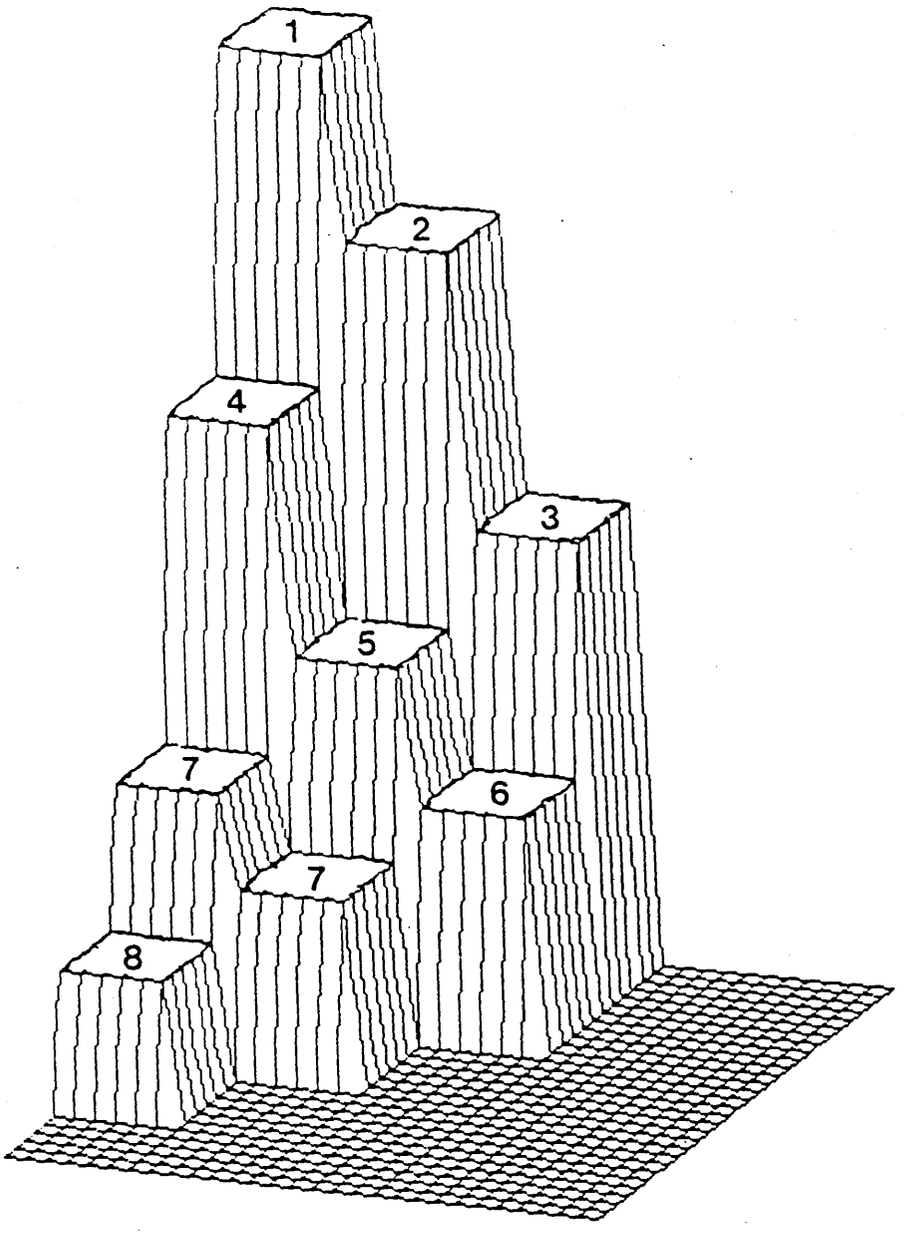

FIGURE 2. - 3 dimensional representation of the mean % Chromium and Fluoride concentrations of sample positions 1-8 (used in figures 3-6).

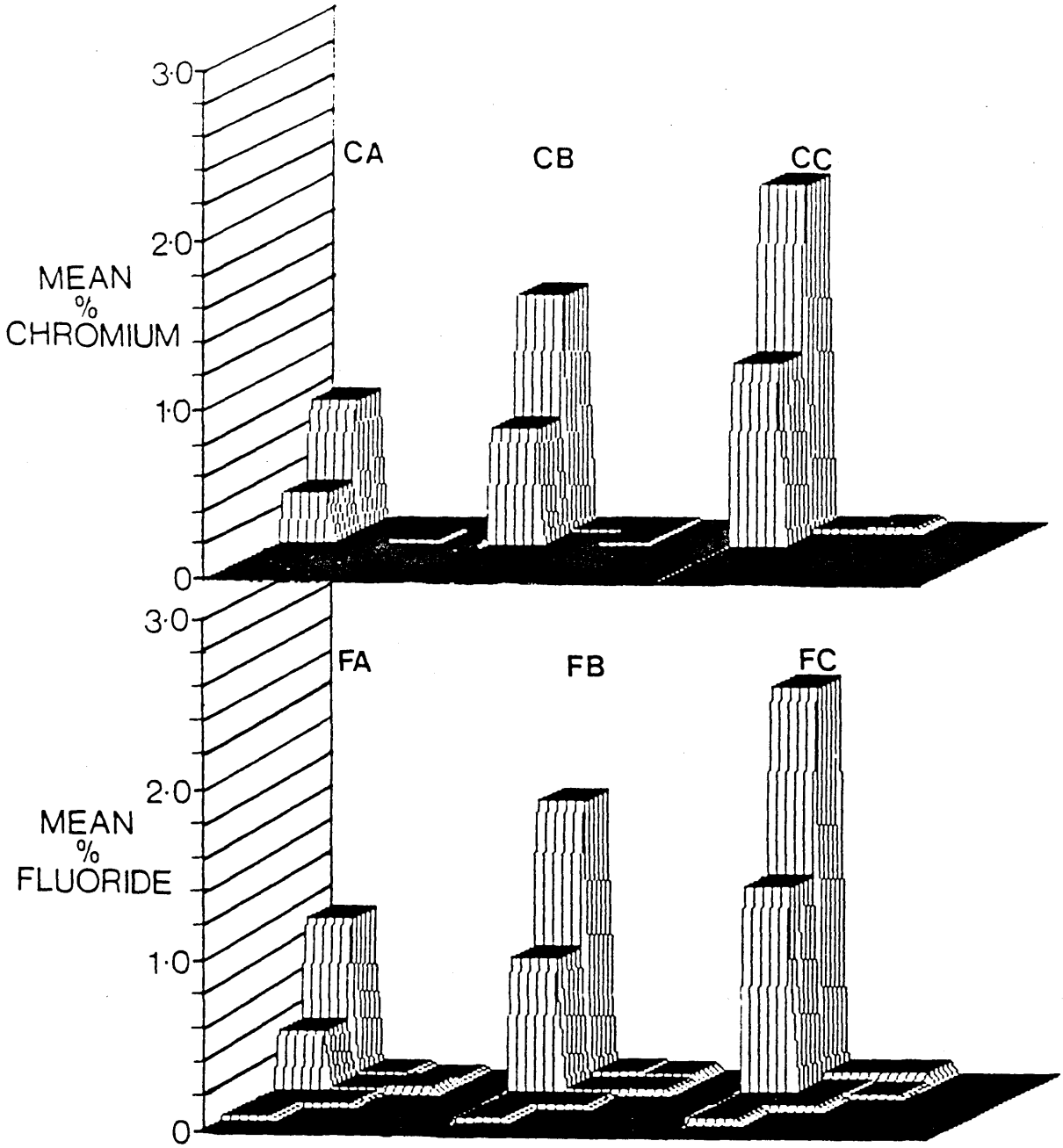

FIGURE 3.

Mean % Chromium (C/A,B,C) and Fluoride (F/A,B,C) concentrations of each sample position 0 months after treatment.
A - 175mm below groundline.
B - Mean values for both sample heights.
C - 175mm above groundline.

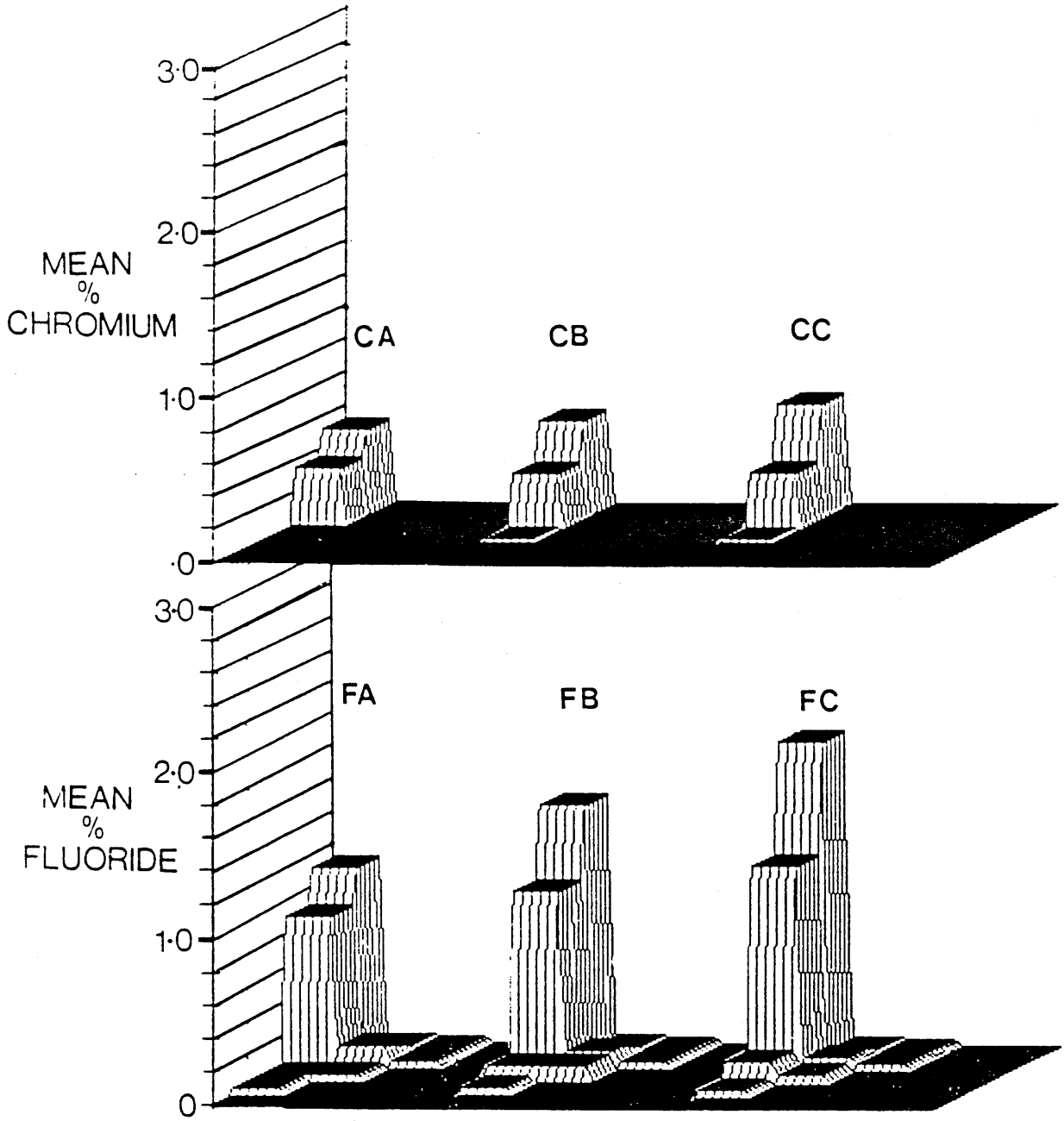

FIGURE 4.

Mean % Chromium (C/A,B,C) and Fluoride (F/A,B,C) concentrations of each sample position 2 months after treatment.
A - 175mm below groundline.
B - Mean values for both sample heights.
C - 175mm above groundline.

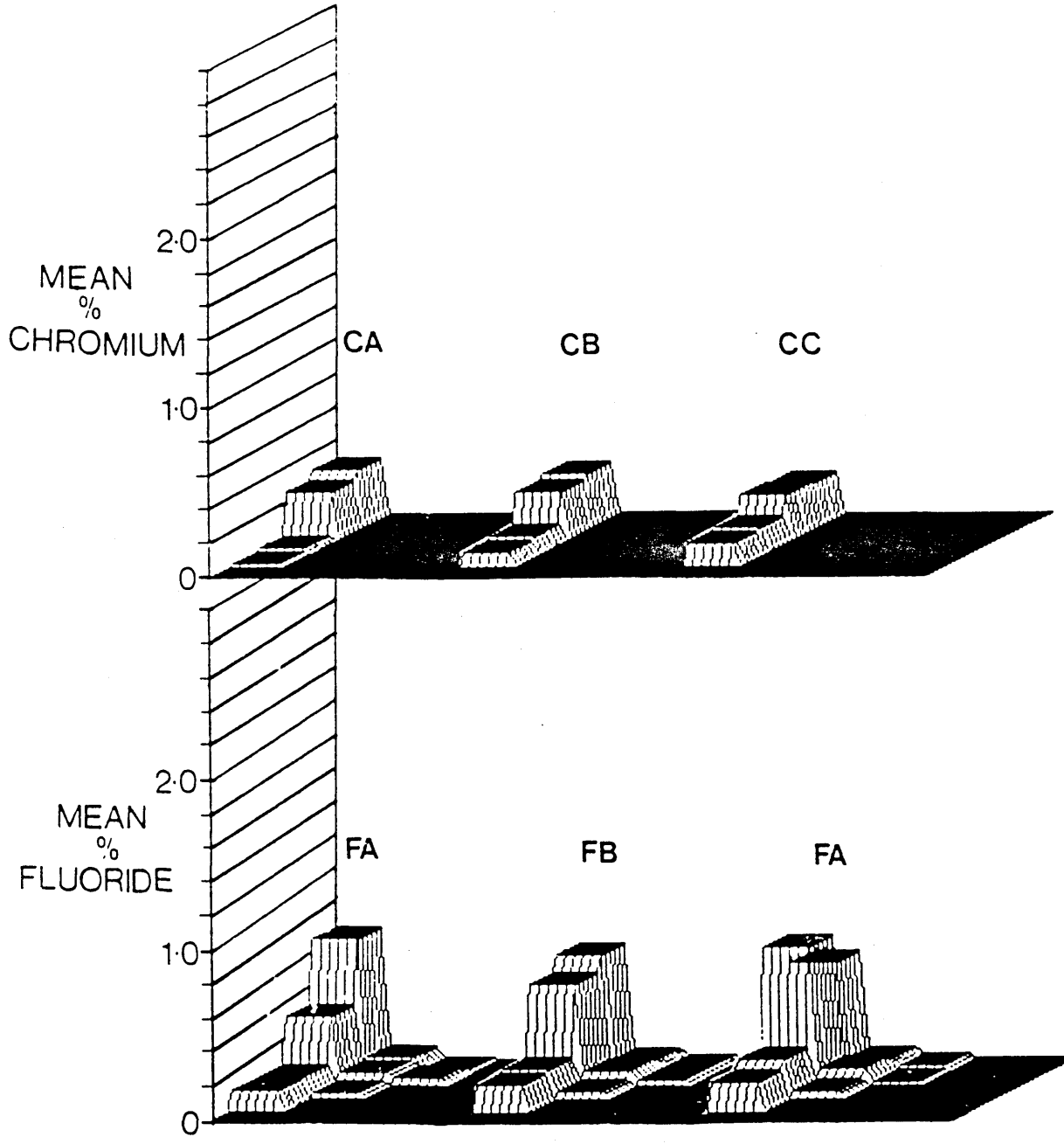

FIGURE 5.

Mean % Chromium (C/A,B,C) and Fluoride (F/A,B,C) concentrations of each sample position 5 months after treatment.
A - 175mm below groundline.
B - Mean values for both sample heights.
C - 175mm above groundline.

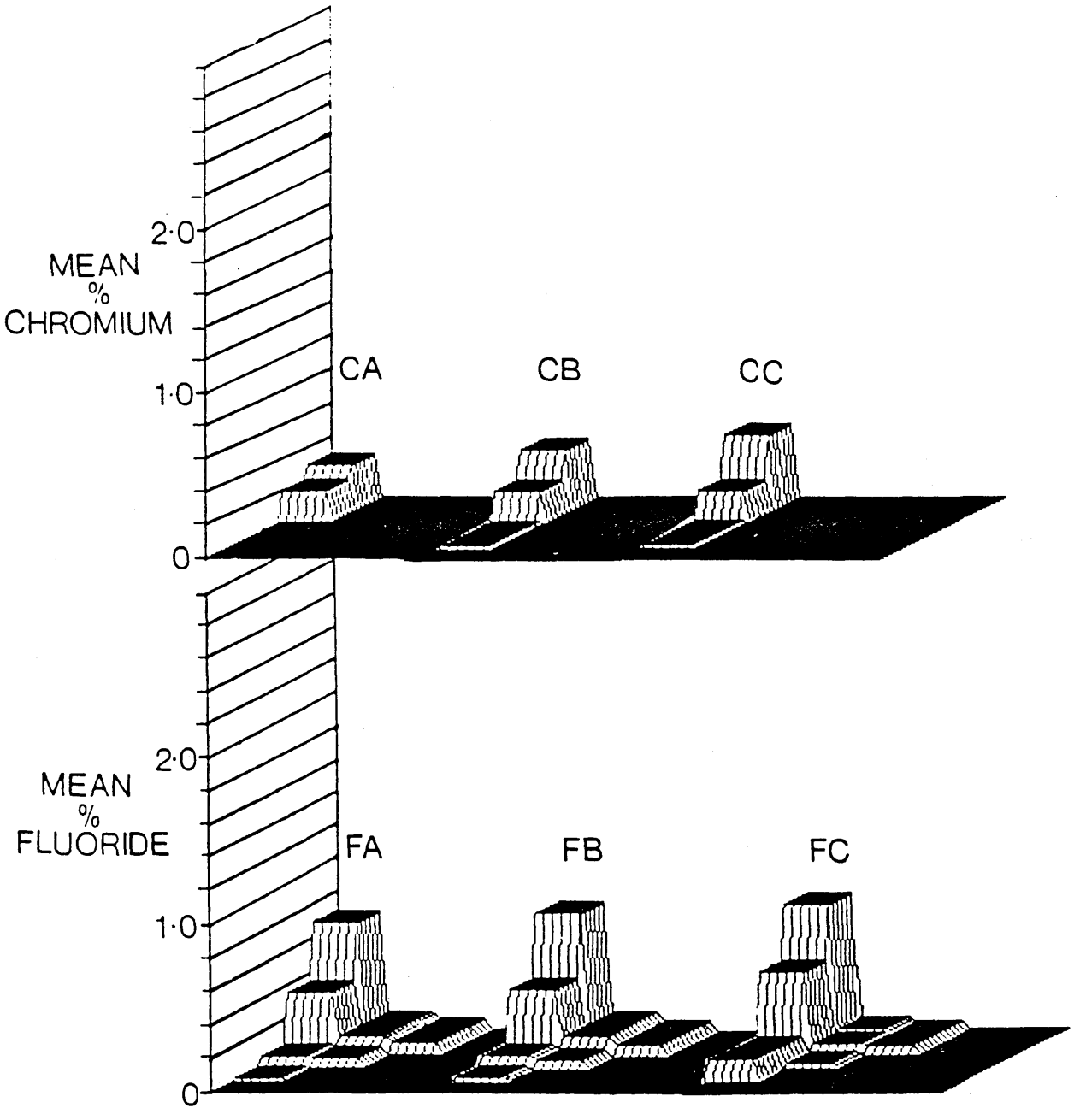

FIGURE 6.

Mean % Chromium (C/A,B,C) and Fluoride (F/A,B,C) concentrations of each sample position 12 months after treatment.
A - 175mm below groundline.
B - Mean values for both sample heights.
C - 175mm above groundline.

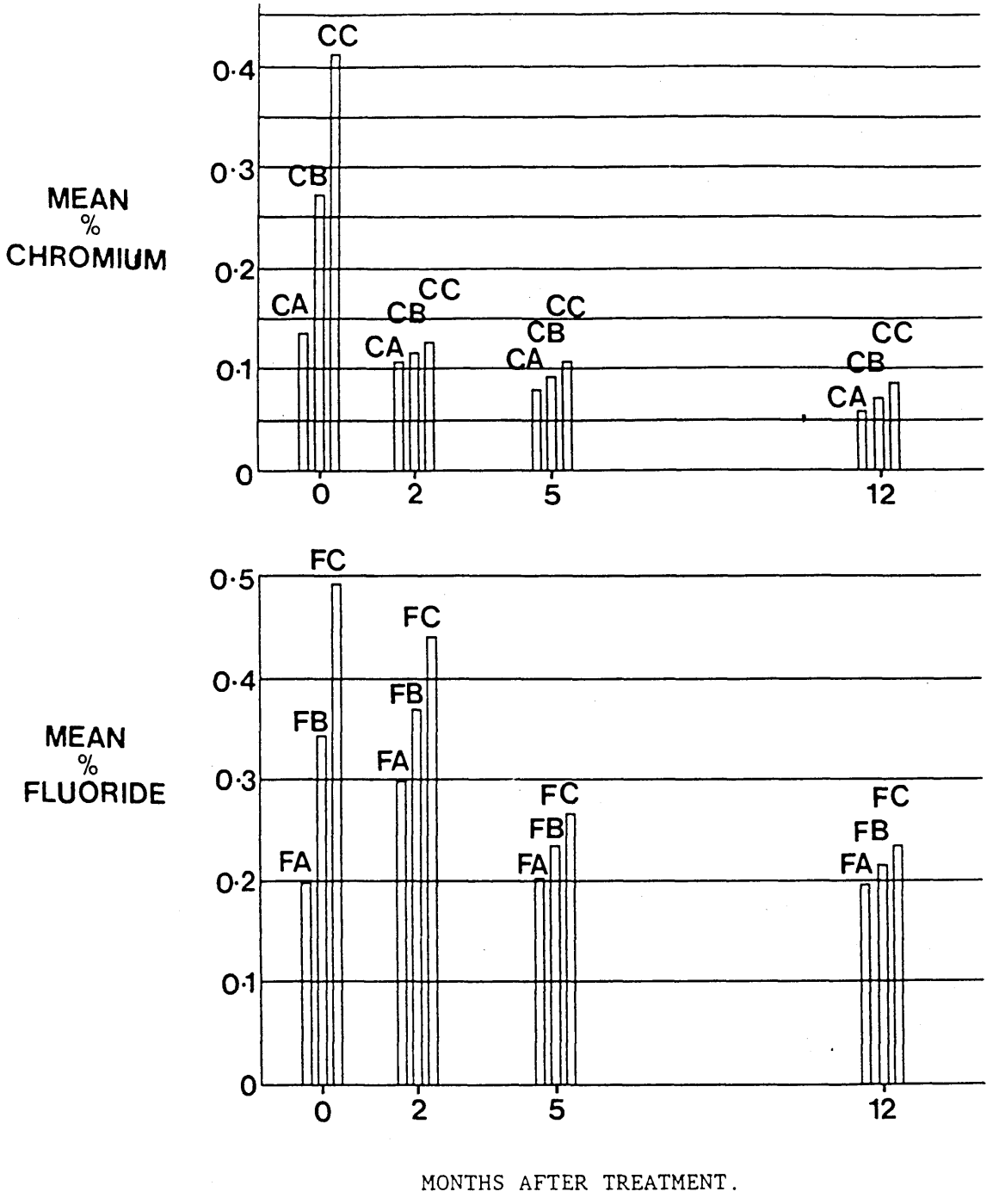

FIGURE 7. - Mean % fluoride (F/A,B,C) and chromium (C/A,B,C) concentrations.
A - 175mm below groundline.
B - Mean value for both sample heights.
C - 175mm above groundline.

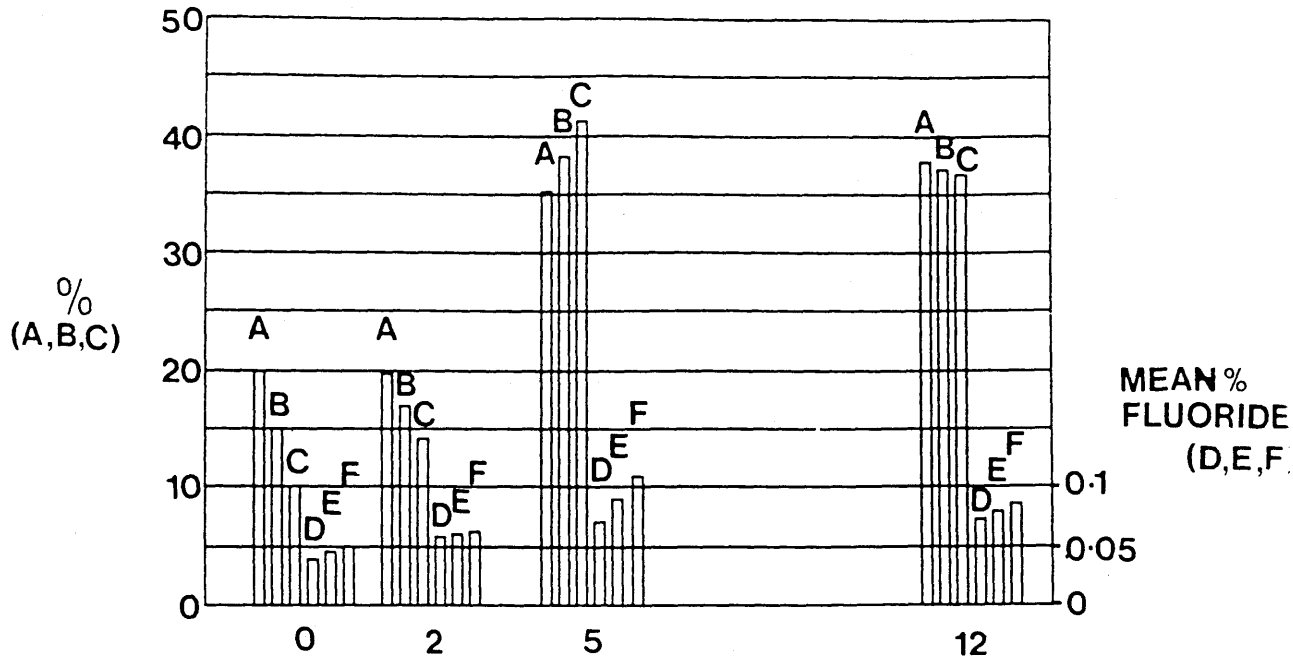

FIGURE 8. - Mean % fluoride concentrations for all sample positions excluding 1 and 4 (D,E,F) and expressed as a percentage of the total mean % fluoride concentration for all sample positions (A,B,C).
A+D - 175mm below groundline.
B+E - Mean value for both sample heights.
C+F - 175mm above groundline.

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Working Group II Fundamentals of Testing.

Development of a model system to assess the efficacy and environmental impact of a chromated fluoride remedial treatment for creosoted distribution poles.

by

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DEVELOPMENT OF A MODEL SYSTEM TO ASSESS THE EFFICACY AND ENVIRONMENTAL IMPACT OF A CHROMATED FLUORIDE REMEDIAL TREATMENT FOR CREOSOTED DISTRIBUTION POLES.

Derek C.R. Sinclair, George M. Smith, Alan Bruce and Bernard King.

ABSTRACT.

A closed model system was designed to facilitate a controlled study of the leachability and environmental fate of a remedial preservative under laboratory conditions. The elements of the model include a precipitation apparatus above a treated pole section which is positioned in a representative soil profile supporting a sward of perennial ryegrass. The model will allow detailed examination of the movement of any toxic preservative constituents, in soil and water, released by an accelerated regime of simulated rainfall. Chemical analysis of soil and leachate will be complimented by plant analysis to identify bioaccumulation of any soil contaminants leached from the treated pole section.

This paper details the design and development of the system from earlier environmental models, the difficulties encountered in construction and the sampling regimes to be employed. The benefits of such a system for inclusion in preservative testing protocols is discussed.

Keywords: Creosoted poles; Rentex; chromium; fluoride; model system; environment; risk assessment.

1. INTRODUCTION.

The preservative oil creosote is extensively used in the United Kingdom as a preservative pre-treatment applied by vacuum/pressure impregnation to electricity distribution poles of Scots pine and affords long term protection from decay. Treatment is limited to the sapwood region of the pole and the unprotected heartwood of the interior can, after a time in service, be subject to decay. Pole failure may result, most commonly at the groundline, where moisture and oxygen conditions are most conducive to the growth of decay fungi.

The electricity supply industry has, for many years, attempted to control this problem by employing waterborne fluoride preservatives as groundline remedial treatments for poles in service (Steinherz 1939, Becker 1973).

The most recent preservative formulation proposed as a remedial treatment for poles is Rentex, a paste containing the water soluble salts sodium fluoride, ammonium bifluoride and sodium dichromate as its active constituents. Application is by forcing preservative into the pole via a hollow injection needle propelled by a mechanical pump. Injections are applied in a defined pattern to the pole in a treatment zone 35 cm above and below the groundline. A bitumen coating is applied to the treated area and an aluminium sheath is placed round the treated area above the groundline (Morris and Calver 1987).

A limited field evaluation of the efficacy of the Rentex treatment has been carried out (Bruce and King 1989), with favourable results. This represented the only work undertaken to establish the suitability of Rentex as a remedial groundline treatment for distribution poles.

In view of this, a project was undertaken to develop valid testing protocols to permit extensive evaluation of the efficacy and environmental impact of the Rentex treatment. Field trials were conducted to establish: preservative diffusion in treated poles;levels of control of decay organisms; long term maintenance of preservative action and leaching and recalcitrance of the preservative constituents in the soil environment adjacent to treated poles.

Chemical analysis of soil samples recovered from around Rentex treated 'on-line' distribution poles and pole sections at a variety of field sites indicated persistent concentrations of fluorine and chromium significantly greater, in statistical terms, than background values. This unpublished data confirms the apparent leach losses of preservative constituents suggested by findings of a simultaneous field study of the diffusion of chromium and fluoride in Rentex treated pole sections (Sinclair et al 1991).

These findings question the long term effectiveness of Rentex treatment and suggest the need for a comprehensive study to identify any environmental effects associated with leached chromium and fluoride components of the preservative.

2. ENVIRONMENTAL HAZARD ASSESSMENT OF CHEMICALS.

A logical 3-stage hazard evaluation of any chemical in the environment has been advocated by Bro-Rasmussen (1988).

Initially the chemical's hazardous properties such as toxicity, persistence and environmental mobility must be identified. The second stage is an assessment of the potential for these properties to be translated into adverse effects on natural systems determined to be at risk due to the specific environmental exposure of the chemical. Finally, an assessment is required as to whether or not this potential is realised by virtue of the intensity, frequency and duration of the chemicals environmental exposure and the actual measured effects on susceptible natural systems identified at stage 2.

3. ENVIRONMENTAL HAZARD ASSESSMENT OF RENTEX.

The leached Rentex constituents, chromium and fluoride, are widely known to display toxicity towards plants and animals (Turner and Rust 1971; Mortvedt and Giordano 1975; Singh et al 1979a, b ; Curtis et al 1979).

In view of significant rises in levels of chromium and fluoride in soil around Rentex treated distribution poles, adjacent plant populations would therefore be worthy of study for toxic effects. The probability of an effect on plant life is supported by the findings of Breeze (1973) and Skeffington et al (1976), which showed that of the 2 main oxidation states of chromium in soil (Cr (III) and (VI)), chromium (VI) the more mobile form, present in Rentex, displayed a greater toxicity to plants. The likelihood of toxic effects due to fluoride is suggested by the reduced yields of rice and wheat found by Singh et al (1979a, b) when additions of fluoride were made to soil.

Increases in soil levels of chromium and fluoride brought about by the use of Rentex are localised around treated poles indicating that toxic effects on animal life via persistent exposure to possibly contaminated plant sources or wind-blown soil particles is unlikely.

Groundwater contamination by leached preservative constituents is more probable. Calder (1988) concluded that chromium (VI) tends to be moderately to highly mobile in most neutral to alkaline groundwaters, and Larsen and Widdowson (1971) found that levels of soluble soil fluoride increased when additions to soil were made. Contamination of groundwater may allow entry of toxic levels of chromium and fluoride into nearby aquatic ecosystems. Evidence of the toxicity of chromium in an aquatic environment is extensive (Anonymous 1976) and fluoride has been shown to be toxic to a number of freshwater organisms (Curtis et al 1979).

The probability that the known toxicity and mobility of chromium and fluoride in soil (Bro-Rasmussen (1988) Hazard Assessment Stage 1) will bring about measurable environmental contamination and damage to plants and groundwater by virtue of persistent elevated levels of these chemicals adjacent to Rentex treated distribution poles (Stage 2), requires confirmation by further study (Stage 3).

To identify, examine and quantify the environmental impact of wood preservatives, or other chemicals, under field conditions with any degree of accuracy and within a reasonable timescale is difficult and expensive. Consequently, many workers have employed model field systems for this purpose.

4. ENVIRONMENTAL IMPACT ASSESSMENT USING MODEL SYSTEMS.

Most model systems to date have been developed to examine the environmental fate of agricultural pesticides (Metcalf 1974; Roberts 1976 and Beall et al 1976). In the field of wood preservation, Wegen (1990) and Murphy and Dickinson (1990) have carried out detailed studies using limited model systems.

Wegen (1990) examined the leaching of a chromium/copper salt wood preservative in water contact. Leaching of preservative constituents decreased as the time interval between wood treatment and water contact increased. Toxicity to fish was proportional to the amount of preservative leached. No account was taken of the dilution effects of a flowing water system or fish migration from the toxic source.

In a study of the leaching of copper chromium arsenic preservatives, Murphy and Dickinson (1990), placed treated wooden stakes in small containers of different soil type and pH. Leaching was carried out by flooding the soil with water of different pH. Chemical analysis indicated that leachate and soils had accumulated preservative components, concentrations varying with experimental conditions and component type. The size and condition of the stakes was unrepresentative of treated field structures and bioassays, necessary to confirm the environmental hazard potential of leached chemicals, were not undertaken.

A representative model field system for accelerated evaluation of the Rentex preservative was designed and assembled, to include the experimental parameters outwith the scope of the experimental designs of Wegen (1990) and Murphy and Dickinson (1990), with reference to the environmental hazard indicators (plants and groundwater) identified for Rentex.

5. MODEL FIELD SYSTEM FOR RENTEX.

The model system consists of a 2m long Rentex treated pole section placed vertically in a grass covered soil bed which contains a number of simulated field drains for leachate collection. An overhead tapwater misting unit is included to provide simulated rainfall and lighting was provided on a day/night cycle. Three models were prepared, 2 containing Rentex treated creosoted pole sections in soils of different texture and a control containing a creosoted pole section untreated with Rentex.

5.1 SOIL PREPARATION.

A free draining sandy loam soil was obtained from the Scottish Crop Research Institute at Dundee. The soil was collected from the upper 15cm of topsoil from a field site which had received no previous chemical treatment, and was stored outside in covered bins till required.

Initially a stoney base for each soil bed was produced by utilising that fraction of the soil failing to pass a 1cm mesh sieve. Soil fractions which passed and failed to pass a $0.5cm^2$ mesh sieve were used as topsoil and subsoil respectively.

Soil profiles were constructed within three 227 litre high density polyethylene water tanks of 55 * 55 * 108cm. The layers of each profile were given a 2° slope to represent prevailing field conditions (figure 1). The topsoil and subsoil of one soil bed were amended by the addition of 1 part washed sand to 2 parts soil by volume.

5.2 CONSTRUCTION OF DRAINS FOR LEACHATE COLLECTION.

During the construction of each soil profile, a number of artificial field drains were positioned at various levels (figure 2). The drains consisted of 12mm bore PVC piping, pierced on the upper surface with nine 4mm diameter holes for every 50mm of length (figure 3). After placing each drain a permeable 30mm deep layer of washed aquarium gravel was added to facilitate water movement and prevent blockage by soil. Figure 4 shows a side elevation of the soil bed detailing drains 1-3 and 6-8. The continuous gravel layer above the lower drains (6-8) was designed to channel a broad front of drainage water from above to these drains to prevent flooding of the soil bed.

5.3 TESTING THE EFFICIENCY OF THE DRAINAGE SYSTEM.

On completion of the soil beds and drainage system, watering of the soil surface was carried out using a standard watering can with rose spout. Watering was continued until soil saturation was identified by constant flow of water from the lower drains 5 and 9 (figure 2). Further watering did not result in flow through drains 1-3 and 6-8 and resistance to free drainage was indicated by surface water accumulation.

The soil beds were excavated and non-functioning drains were found to be blocked by soil of a muddy consistency. Soil structure, weakened by the necessary removal of large stones, earth clods and plant material during sieving, had broken down. Accordingly all topsoil and subsoil was removed and air dried to a moist consistency. The structure and free draining nature of the soil was re-introduced by a one third addition by volume of washed aquarium gravel. Soil profiles and drains were re-constructed and on testing, a free flow of water through all drains was achieved.

5.4 SELECTION AND TESTING OF A GRASS VARIETY FOR INCLUSION IN THE MODEL.

Perennial ryegrass (Lolium perenne) was chosen as a bioassay for its reported sensitivity to chromium (Breeze 1973) and dominant presence adjacent to 'on-line' distribution poles used in field trials carried out as part of this overall assessment of Rentex.

Seed for the perennial ryegrass variety 'Hunter' was obtained from the Scottish Crop Research Institute at Dundee. To evaluate its suitability, all soil beds were handsown at a seeding rate of $90g/m^2$. This heavy seeding rate was designed to ensure a uniform density of growth over each surface, removing plant density as an experimental variable.

Once each soil bed was seeded, illumination was provided on a day/night cycle of 14/10 hours by a Complex plantcare 160W Mercury Fluorescent Plant Irradiator positioned 90cm above each soil surface. Temperature and relative humidity were measured at the soil surface and monitored using a Vaisala HM 34 Humidity and Temperature meter.

One week after sowing, a uniform emergence of young shoots was obtained and after 1 month a dense young sward was established (figure 5). At 6 weeks after sowing a general chlorosis was observed within the grass canopy, which progressed rapidly till widespread shoot death occurred.

Subsequently, various regimes of sowing, watering and cutting were tested to identify the measurement limitations this variety of ryegrass would impose on the model. Initial emergence and growth of 'Hunter' was excellent, but plant death always occurred at 6 to 7 weeks after sowing due to the collapse of the grass canopy and consequent rotting of tillers leading to root death as photosynthate was denied. This could not be prevented by cutting to encourage younger more erect tillers as this resulted in 'die-back'.

It was concluded that this grass variety was unsuitable for the length of trial envisaged. The perennial ryegrass variety 'Fennema' was therefore obtained from Twyford Seeds Limited, it's characteristics of good ground cover and proliferation on cutting being ideal.

The remaining growth of the 'Hunter' variety was cut to encourage 'die-back' and germination of seedlings from unsprouted seed, which were also subsequently cut. The soils were therefore cleared for the 'Fennema' variety which was sown at a rate of 90g/m ten days after Rentex treated and control pole sections were placed in each soil bed.

5.5 POLE SECTION PREPARATION AND INSERTION INTO SOIL BEDS.

Six aged creosoted pole sections of Scots pine were obtained from Hydro Electric Plc at Dundee. Four 2m long sections of equal diameter were Rentex treated by the standard method, each receiving the same number of injections. Bitumen was applied to the treated area and an aluminium sheath attached above the groundline.

All pole sections were stored at a mean relative humidity and temperature of 94% and 24 C respectively to prevent moisture loss prior to use in the model system. After this conditioning period, 2 treated and 1 control pole section were dug into the top of the soil bed slopes, the base of each resting above drain 5 (figure 6). The soil bed amended with sand received one of the treated sections. The remaining pole sections (2 treated and 1 control) were set aside for comparative chemical analysis at the end of the experimental period

A feature of the model is the ease with which the watertable level can be altered to facilitate leaching studies. For this study, the soil bed pole sections were subjected to 2 weeks in contact with a raised watertable, achieved by inverting drains 5 and 9 (figure 6) and watering the soil bed and pole with an overhead tapwater misting unit until the outlets of these drains were full.

5.6 OVERHEAD PRECIPITATION UNIT FOR ACCELERATED LEACHING.

Simulated rainfall for each pole section will be provided by an overhead tapwater misting unit supplied by Philip Harris Education. Three units were connected in series with polythene pipework (10mm diameter), each consisting of an atomiser jet supported 165cm and 15cm above each soil bed and pole top respectively, by a rigid PVC rise pipe (13mm diameter).

Annual rainfall records for 1980-90 at a Rentex field trial site indicated an annual mean rainfall of 1246mm or 104mm monthly. A simulated rainfall of 52mm every third day was therefore chosen as the rainfall regime for the test to provide the mean annual rainfall in 72 days.

Each atomiser jet gave a precipitation coverage of $1m^2$ over each pole and adjacent sloping soil surface. The mean flow rate of 0.367 litres/minute through each jet was established by collecting the flow during a series of timed tests. This flow rate and the need to provide 52mm of simulated rainfall over 1m indicated that each spray jet would require to operate for 140 minutes at each rainfall simulation.

6. SAMPLING OF THE MODEL SYSTEM.

Samples of leachate, plant material, soil and treated wood will be collected for analysis over the experimental period. Chemical analysis for fluoride and total chromium will be based on a fluoride analysis method developed by Mcquaker and Gurney (1977) and modified by Sinclair et al (1991).

Samples of leachate will be collected via the drainage system after each application of 'rainfall' for pH measurement and chemical analysis of total chromium, chromium (VI) and fluoride content. This will identify differences in movement of leached preservative constituents to the watertable (drains 5 and 6), through the soil profile (drains 1-3 and 6-8) and in any surface 'run-off' waters (drain 4).

Plant shoot and root samples will be collected for chemical analysis and growth measurement, during and on completion of the experiment, immediately downslope of the pole section in an area 30cm * 60cm long. The sampling system adopted ensures that plants from the same position, relative to the pole, will be taken at each time interval.

Soil removed from the plant root samples will be analysed for total chromium and fluoride content to determine any relationship between soil concentrations of these chemicals and plant uptake. Soil samples from greater depth will be recovered for chemical analysis at the end of the experiment.

On completion of analysis of all soil, plant and leachate samples representative wood samples from the leached and unleached treated pole sections will be analysed to quantify preservative loss to the system. In addition, wood samples will be recovered according to the procedure of Sinclair et al (1991) and analysed to examine diffusion of chromium and fluoride within the Rentex treated pole sections.

7. ASSESSMENT OF THE MODEL SYSTEM.

The challenge in designing accelerated model systems to evaluate environmental impact of wood preservatives is to provide conditions which allow critical comparisons with field situations. A number of measurements from the system described above will compliment field data of the behaviour of the preservative Rentex (e.g. chemical analysis of soil and wood). This will permit comparative statistical analysis to verify the representative nature of the model and the accuracy of any environmental effects identified.

The model will partially fulfil the requirements of the hazard assessment method advocated by Bro-Rasmussen (1988) by determining the nature and intensity of any environmental impact on plants and groundwater caused by elevated soil levels of chromium and fluoride from the normal field exposure of distribution poles remedially treated with the preservative Rentex.

If the findings from the model system indicate groundwater contamination with chromium and fluoride from Rentex treated poles further development will be required to evaluate possible harmful effects on aquatic ecosystems.

The design of the model system lends itself to adaption for hazard assessment of other wood preservatives and chemicals in the environment. It would therefore represent a useful addition to many preservative testing protocols.

ACKNOWLEDGEMENTS.

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FIGURE.1 Exposed section of soil tanks showing sloping soil profiles constructed from sieved fractions of original field topsoil.

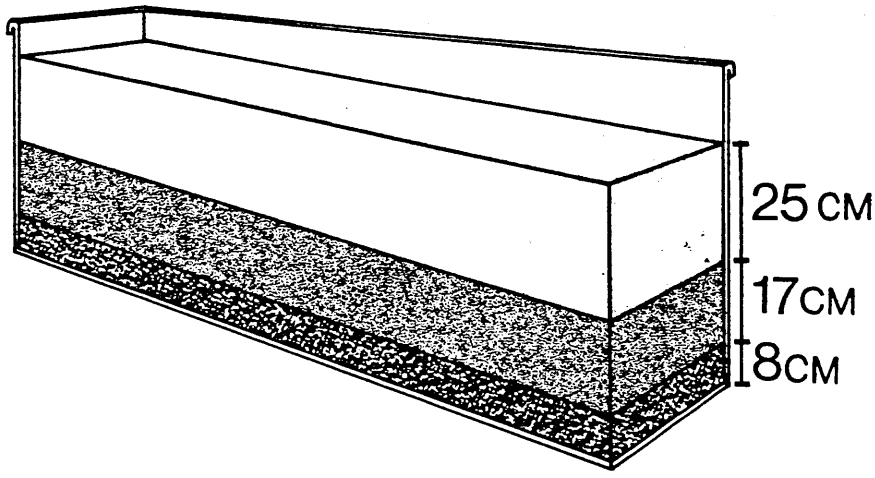


FIGURE.2 Arrangement and numbering system of drains within soil profile for leachate collection.

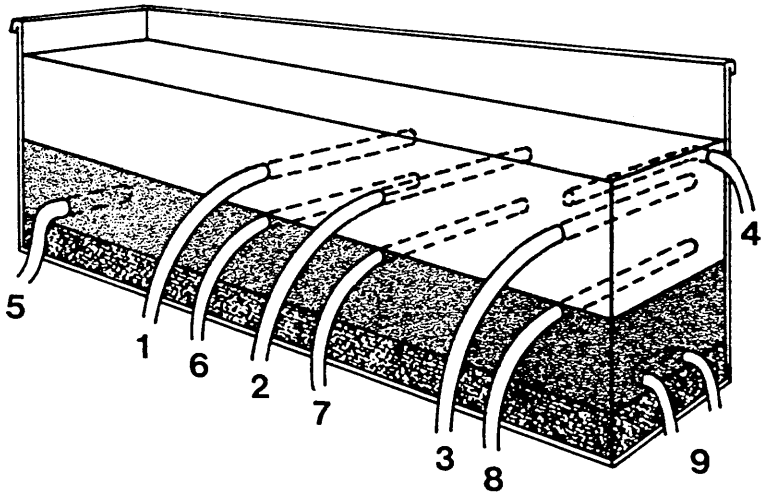


FIGURE.3 Detail of PVC piping drain with holes to receive water flow from surrounding soil.

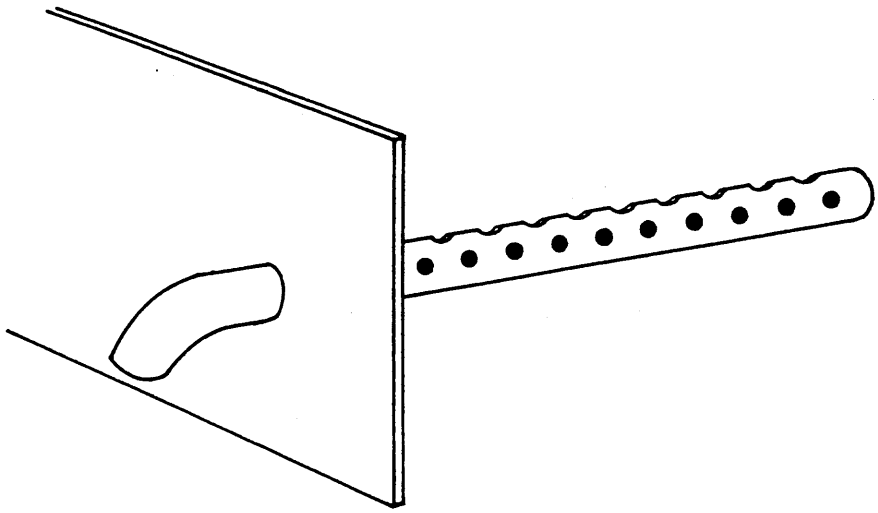


FIGURE.4 Side elevation of soil profile drains detailing drainage holes and permeable gravel layers.

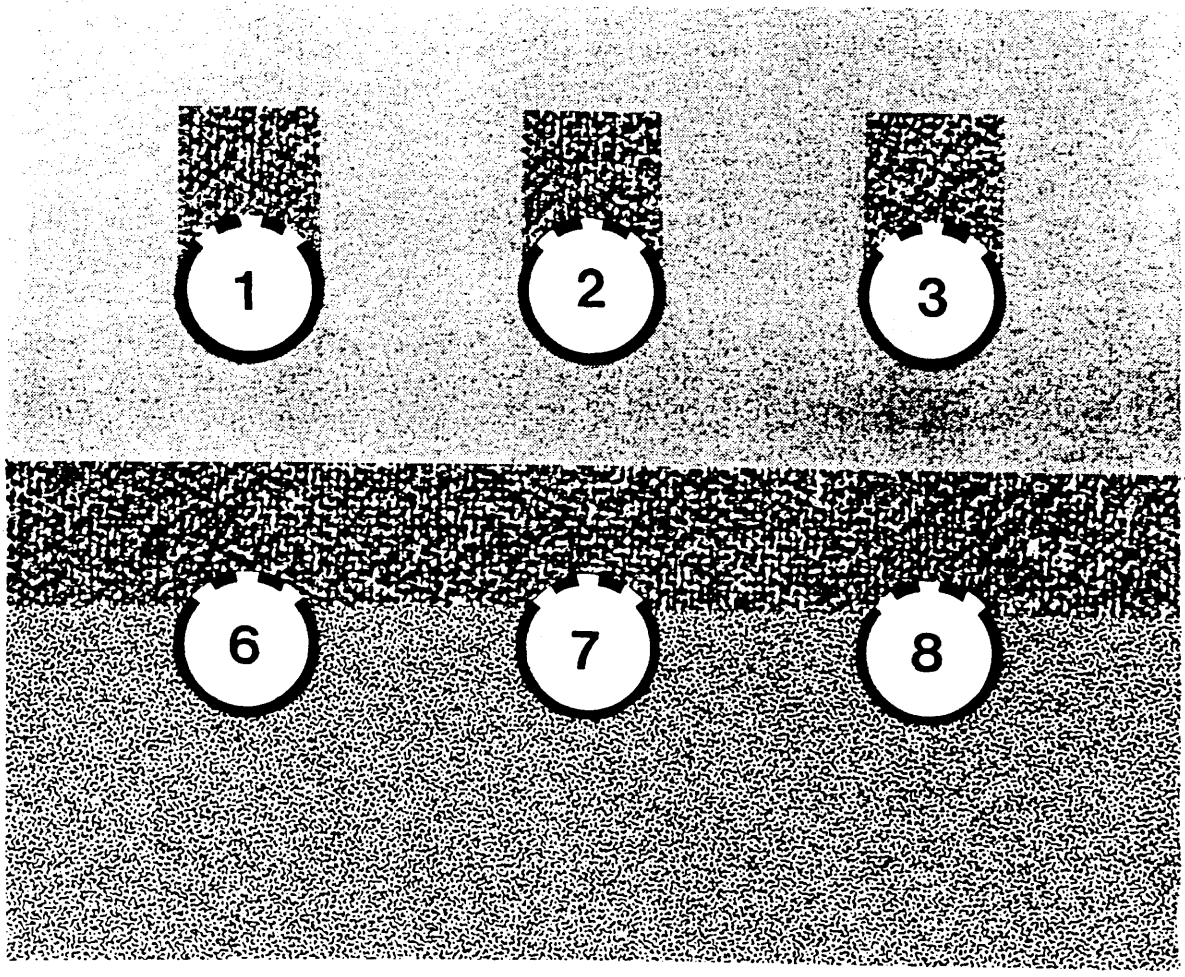


FIGURE.5 Established sward of perennial ryegrass on surface of soil bed.

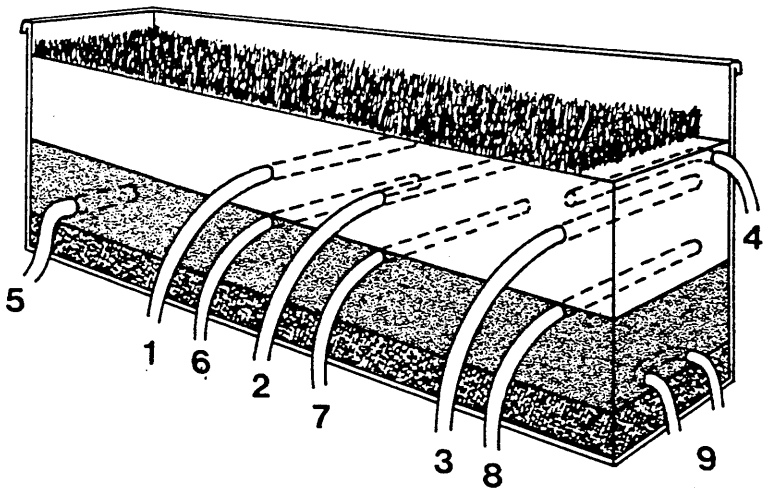
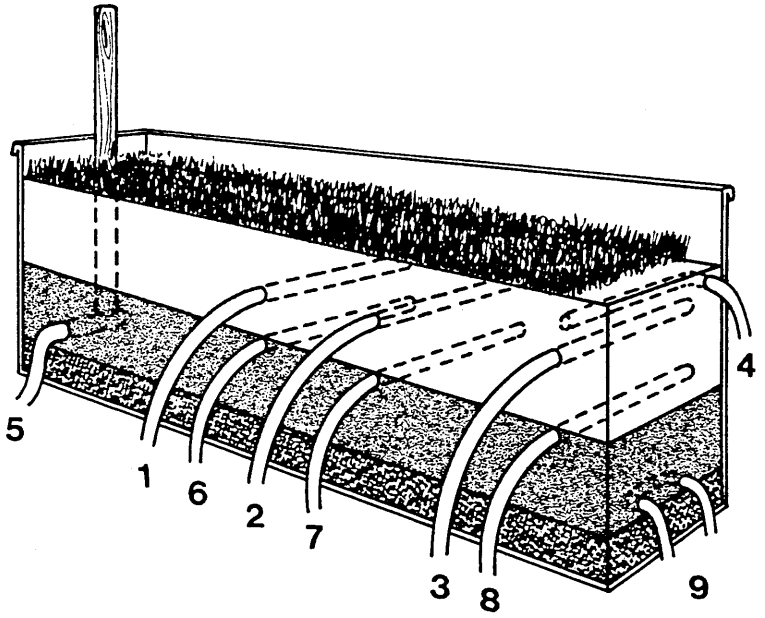


FIGURE.6 Treated pole section in place at top of sloping soil bed.



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Initial Results and Observations of a Model System to Assess the Efficacy and Environmental Impact of Preservative Treated Wood

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INITIAL RESULTS AND OBSERVATIONS OF A MODEL SYSTEM TO ASSESS THE EFFICACY AND ENVIRONMENTAL IMPACT OF PRESERVATIVE TREATED WOOD.

Derek C.R. Sinclair, George M. Smith, Alan Bruce and Harry J. Staines.

ABSTRACT.

The development of a closed model system for the laboratory assessment of the efficacy and environmental impact of a chromated fluoride remedial treatment for creosoted distribution poles has been described (IRG/WP/2395-92).

The model consists of a precipitation apparatus above a treated pole section positioned in a soil profile from which leachate was collected via a series of simulated field drains. Chemical analyses of leachate and soil provided data indicating movement of toxic preservative constituents from the treated pole section to the model environment. These data were complemented by physical and chemical analysis of a sward of perennial ryegrass supported by the soil profile.

This paper reports initial results and observations in terms of the models' suitability for assessment purposes. The advantages of the model system over traditional field studies are discussed.

Keywords: Creosoted poles; remedial treatments; model system; fluoride; chromium; environmental impact.

1. INTRODUCTION.

The electricity supply industry has for many years employed waterborne fluoride preservatives as a remedial treatment for creosoted distribution poles in service (Steinherz 1939, Becker 1973) to control the growth of decay fungi in the groundline heartwood region which is unprotected by creosote pre-treatment.

Rentex, a paste containing the water soluble salts sodium fluoride, ammonium bifluoride and sodium dichromate as its active constituents is a recent preservative formulation proposed as a remedial treatment. Preservative is injected in a defined pattern to the pole in a treatment zone 35 cm above and below the groundline. A bitumen coating is applied to the treated area and an aluminium sheath is placed round the treated area above the groundline (Morris and Calver 1987).

Field evaluations of the efficacy of the Rentex treatment have been carried out (Bruce and King 1989, Sinclair et al 1991) with favourable results though indications of leaching of preservative constituents (Sinclair et al 1991) have been confirmed by chemical analysis of soil samples recovered from around Rentex treated 'on-line' poles at a variety of field sites (unpublished). These findings indicated persistent concentrations of fluorine and chromium significantly greater, in statistical terms, than background values at sample positions close to the treated poles.

These findings suggested the need for a comprehensive study to identify any environmental effects associated with leached chromium and fluoride components of the preservative. The known toxicity of fluorine and chromium towards plant life and their likelihood of entry into groundwater supplies advocated the examination of both of these natural systems in any environmental impact assessment of the Rentex treatment. This study was undertaken using a closed model system under laboratory conditions (Sinclair et al 1992).

2. MODEL ENVIRONMENTAL SYSTEM.

The model system consists of a 2m long Rentex treated pole section placed vertically in a grass covered soil bed which contains a number of simulated field drains for leachate collection. An overhead tapwater spray unit is included to provide simulated rainfall and lighting was provided on a day/night cycle. Three models were prepared, 2 containing Rentex treated creosoted pole sections in soils of different texture and a control containing a creosoted pole section untreated with Rentex.

A free draining sandy loam soil was collected from the upper 15cm of topsoil from a field site which had received no previous chemical treatment. A stoney base for each soil bed was produced by utilising that fraction of the soil failing to pass a 1cm mesh sieve. Soil fractions which passed and failed to pass a 0.5cm mesh sieve were used as topsoil and subsoil respectively after receiving a one third addition by volume of washed aquarium gravel to return some soil structure lost on sieving. The topsoil and subsoil of one soil bed were further amended by addition of 1 part washed sand to 2 parts soil by volume. Sloping soil profiles were constructed within three 227 litre polyethylene water tanks of size 55 * 55 * 108 cm (figure 1).

During the construction of each soil profile, a number of artificial field drains were positioned at various levels (figure 2). The drains consisted of 12mm bore PVC piping, pierced on the upper surface with nine 4mm diameter holes for every 50mm of length (figure 3). After positioning each drain a permeable 30mm deep layer of washed aquarium gravel was added to facilitate water movement and prevent blockage by soil. Figure 4 shows a side elevation of the soil bed detailing drains 1-3 and 6-8. The continuous gravel layer above the lower drains (6-8) was designed to channel a broad front of drainage water from above to these drains to prevent flooding of the soil bed.

Perennial ryegrass (Lolium perenne) was chosen as a bioassay for its reported sensitivity to chromium (Breeze 1973) and regular presence adjacent to 'on-line' distribution poles used in field trials carried out as part of this overall assessment of Rentex. Illumination was provided on a day/night cycle of 14/10 hours by a Complex plantcare 160W Mercury Fluorescent Plant Irradiator positioned 90cm above each soil surface. Temperature and relative humidity were measured at the soil surface and monitored using a Vaisala HM 34 Humidity and Temperature meter. The perennial ryegrass variety 'Fennema' was sown at a rate of 90g/m (after a period of suitability testing) ten days after Rentex treated and control pole sections were placed in each soil bed (figure 6). The heavy sowing rate ensured a uniform density of growth over each soil surface removing plant density as an experimental variable.

FIGURE.1 Exposed section of soil tanks showing sloping soil profiles constructed from sieved fractions of original field topsoil.

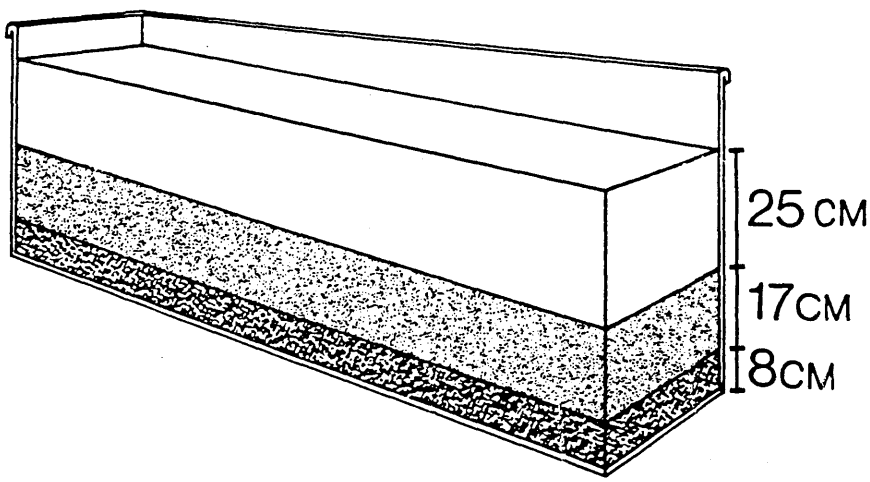


FIGURE.2 Arrangement and numbering system of drains within soil profile for leachate collection.

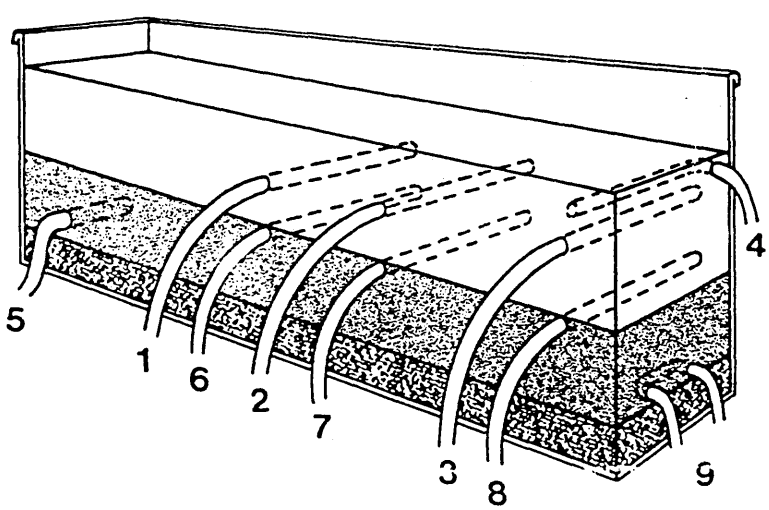


FIGURE.3 Detail of PVC piping drain with holes to receive water flow from surrounding soil.

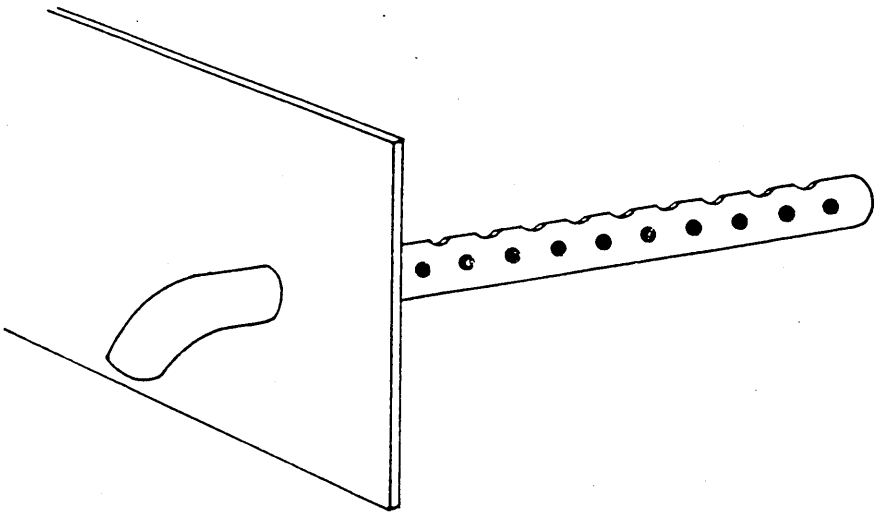


FIGURE.4 Side elevation of soil profile drains detailing drainage holes and permeable gravel layers.

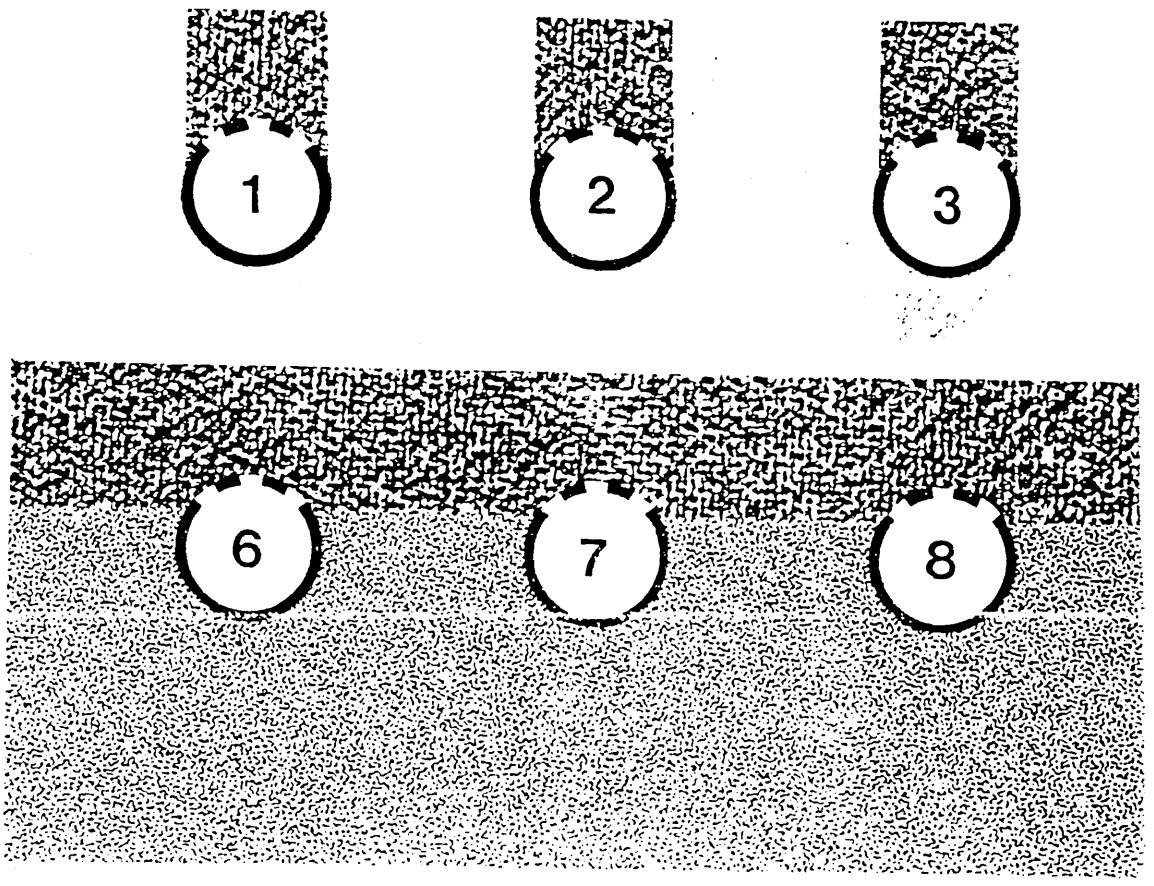


FIGURE.5 Established sward of perennial ryegrass on surface of soil bed.

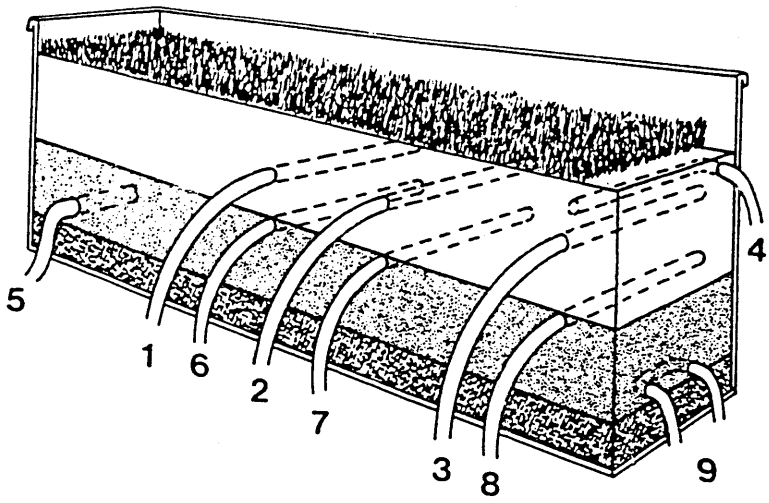
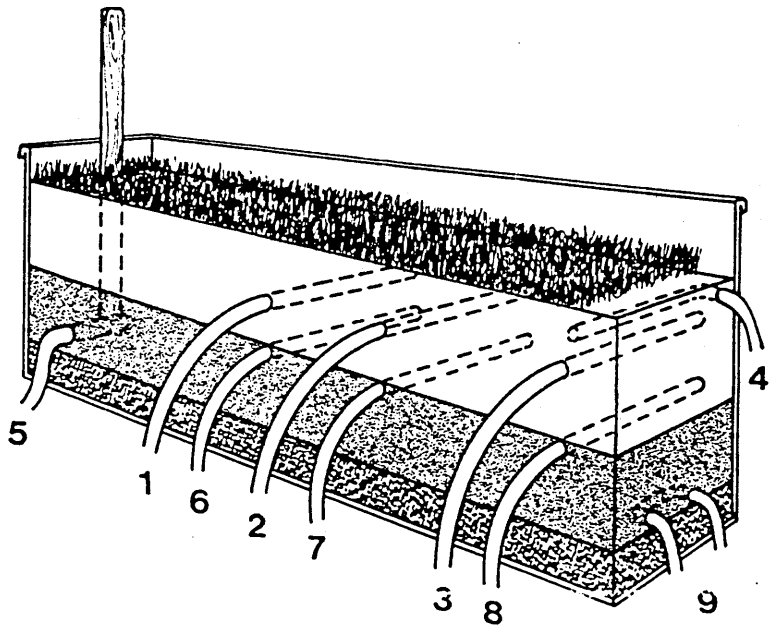


FIGURE.6 Treated pole section in place at top of sloping soil bed.



Six aged creosoted pole sections of Scots pine were obtained and four 2m long sections of equal diameter were Rentex treated by the standard method, each receiving the same number of injections. Bitumen was applied to the treated area and an aluminium sheath attached above the groundline. Two treated and 1 control pole section were dug into the top of the soil bed slopes, the base of each resting above drain 5 (figure 6). The soil bed amended with sand received one of the treated sections. The remaining pole sections (2 treated and 1 control) were set aside for comparative chemical analysis at the end of the experimental period The soil bed pole sections were subjected to 2 weeks in contact with a raised watertable, achieved by inverting drains 5 and 9 (figure 6) and watering the soil bed and pole with an overhead tapwater misting unit until the outlets of these drains were full.

Simulated rainfall for each pole section was provided by an overhead tapwater misting unit supplied by Philip Harris Education. Three units were connected in series each consisting of an atomiser jet supported 165cm and 15cm above each soil bed and pole top respectively. Annual rainfall records for 1980-90 at a Rentex field trial site were consulted to provide an accelerated rainfall regime over each pole and adjacent soil surface. Nine applications of 'rainfall' were carried out accompanied by nine drainage samplings over the experimental period.

3. SAMPLING OF THE MODEL SYSTEM.

Samples of leachate, plant material, soil and treated wood were collected for analysis. Chemical analysis of wood, soil and plant samples for fluoride and total chromium was based on a fluoride analysis method developed by Mcquaker and Gurney (1977) and modified by Sinclair et al (1991).

Leachate samples were collected via the drainage system after each application of 'rainfall' for pH measurement and chemical analysis of total chromium, chromium (VI) and fluoride content.

Plant shoot and root samples were collected for chemical analysis and growth measurement, during and on completion of the experiment, immediately downslope of the pole section.

Soil samples were recovered for chemical analysis at the end of the experiment.

On completion of collection of all soil, plant and leachate samples representative wood samples from the leached and unleached treated pole sections were analysed to quantify preservative loss to the system. In addition, wood samples were recovered according to the procedure of Sinclair et al (1991) and analysed to examine diffusion of chromium and fluoride within the Rentex treated pole sections.

This paper presents analysis data on the leachate samples collected from the model system.

4. ANALYSIS OF LEACHATE.

Measurements of pH were carried out using a Corning Eel model 12 pH meter and electrode.

Fluoride measurements were made using a digital volt-meter equipped with a Russell model 94-4099 fluoride electrode and reference electrode type 900019. Total ionic strength adjustment buffer (TISAB) consisted of 58ml of glacial acetic acid and 12g of sodium citrate dissolved in 300ml of distilled water adjusted to pH 5.2 using 5M NaOH, made up to 1 litre. All reagents used were of analar quality.

Total chromium measurements were carried out using a Perkin Elmer 1100B atomic absorption spectrophotometer.

Determination of chromium (VI), using a colorimetric method developed from analytical techniques detailed by Charlot (1964), was carried out using a Perkin Elmer UV/VIS Spectrophotometer Lambda 2.

5. RESULTS.

Figure 7. - Table of soil characteristics.

	Sandy Loam Soil.	Sand Amended/Sandy Loam Soil.
pH.	6.15	5.45
Cation Exchange Capacity.	11.61	5.96
Organic Matter (g/kg).	33.6	18.8
Water Holding Capacity (%).	18.21	13.75

Figure 8A indicates the amount of simulated rainfall entering each model profile. At 13 days after watering began, the volume of 'rainfall' was increased by 50% to 46.33 litres to further accelerate the leaching regime. This was maintained until a total of 9 waterings over 40 days had been completed, representing half the annual rainfall from a Rentex field trial site. Over the experimental period, each model system containing a pole section received 370.5 litres of simulated rainfall.

It was found that drains 1, 2, 3 and 4 of all 3 models (figure 6) failed to flow due to lack of saturation of soil in the upper profile and the difference in drain design compared to drains 6, 7 and 8 (figure 4). A clay soil would probably have produced a flow through these drains due to its greater moisture holding capacity and resistance to vertical flow of drainage water.

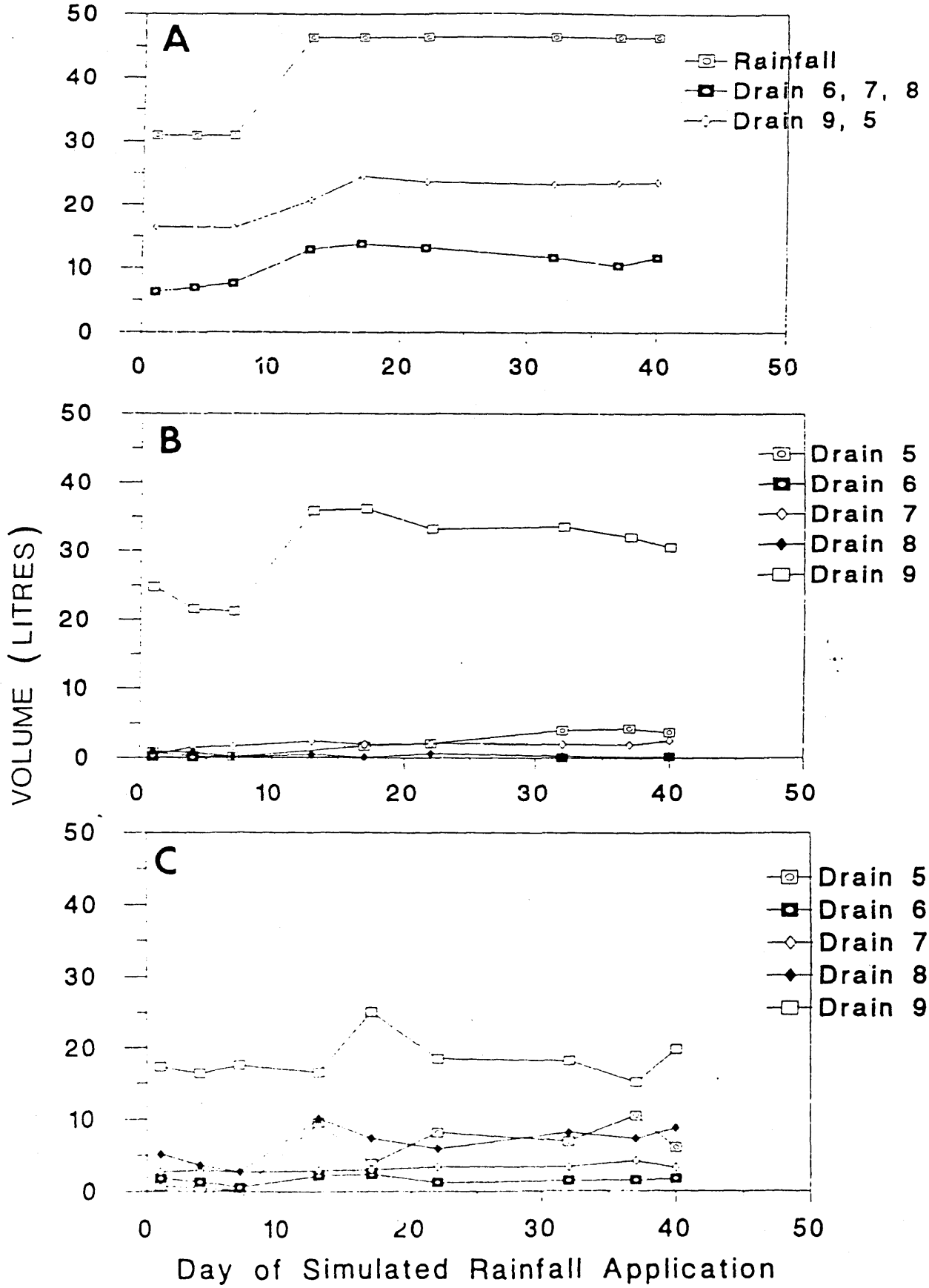


FIGURE - Volume of water in -
A - Simulated rainfall and Control soil leachates.
B - Sand amended treated soil leachates
C - Treated soil leachates

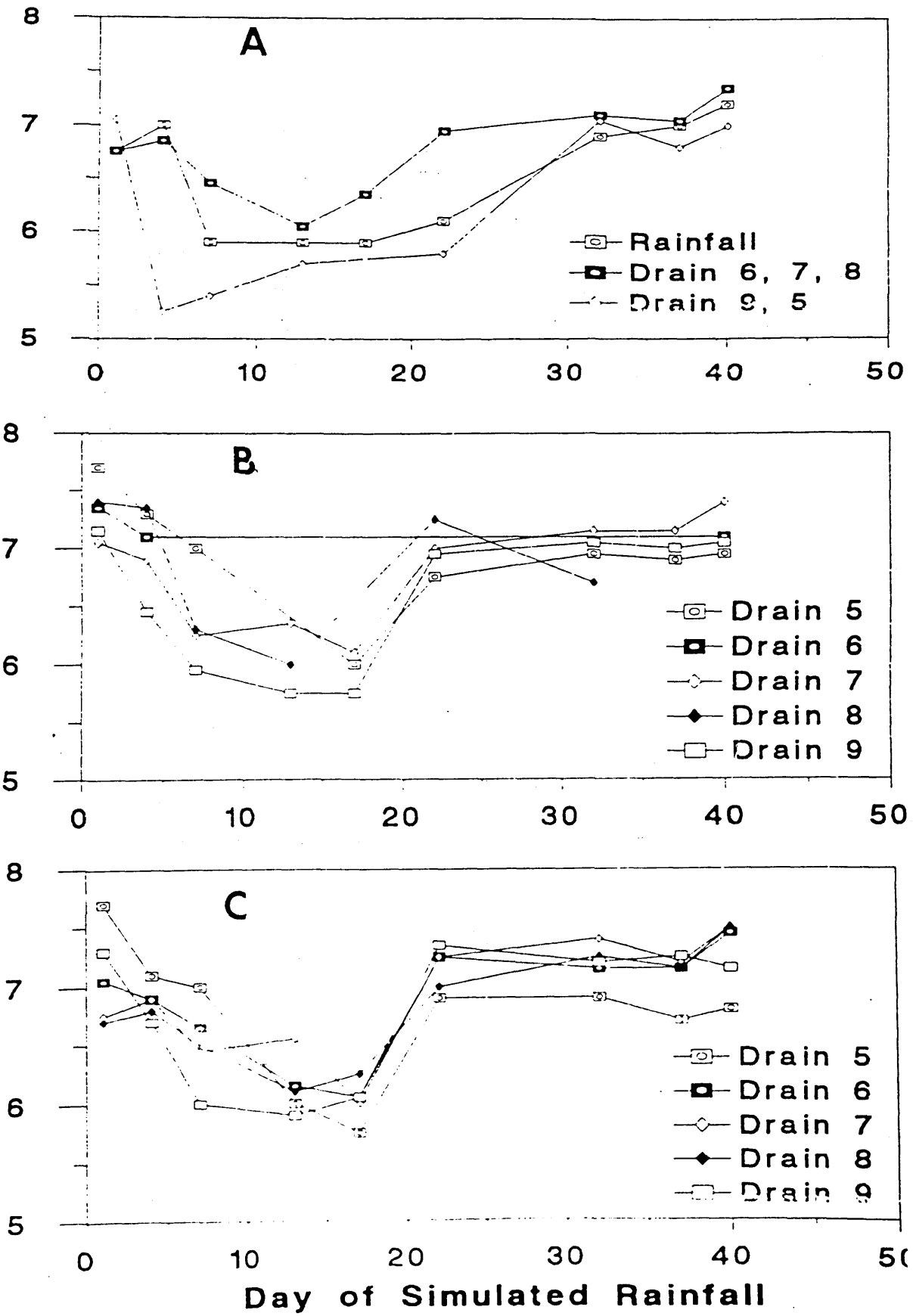


FIGURE 9 pH of : A - Simulated rainfall and control soil leachates.
B - Sand amended treated soil leachates.
C - Treated soil leachates.

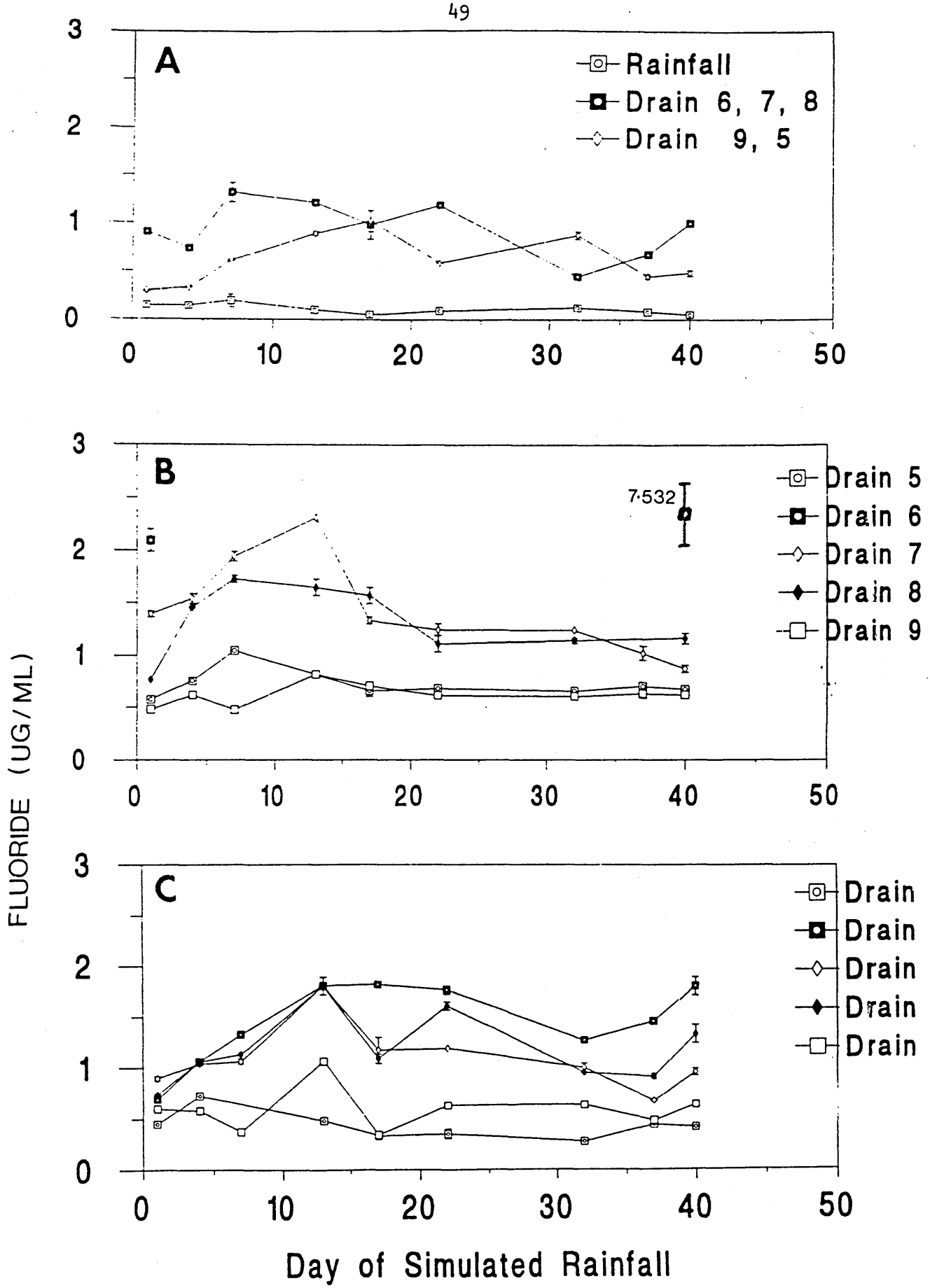


FIGURE.10 Mean Fluoride concentration (ug/ml) of :-
A - Simulated rainfall and Control soil leachates.
B - Sand amended/treated soil leachates.
C - Treated soil leachates.

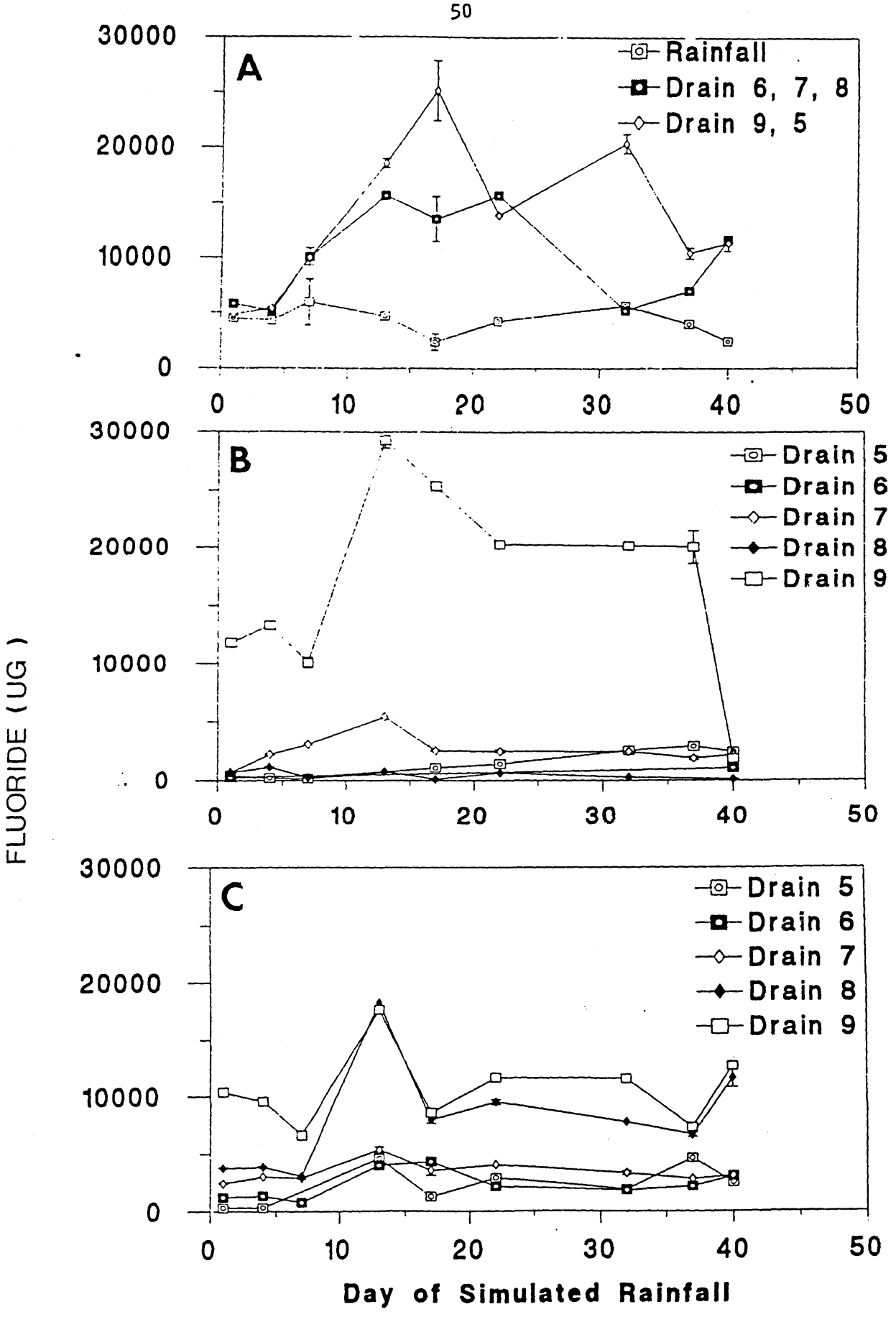


FIGURE.11 Mean bulk Fluoride content (ug * ml) of :-
A - Simulated rainfall and Control soil leachates.
B - Sand amended/treated soil leachates.
C - Treated soil leachates.

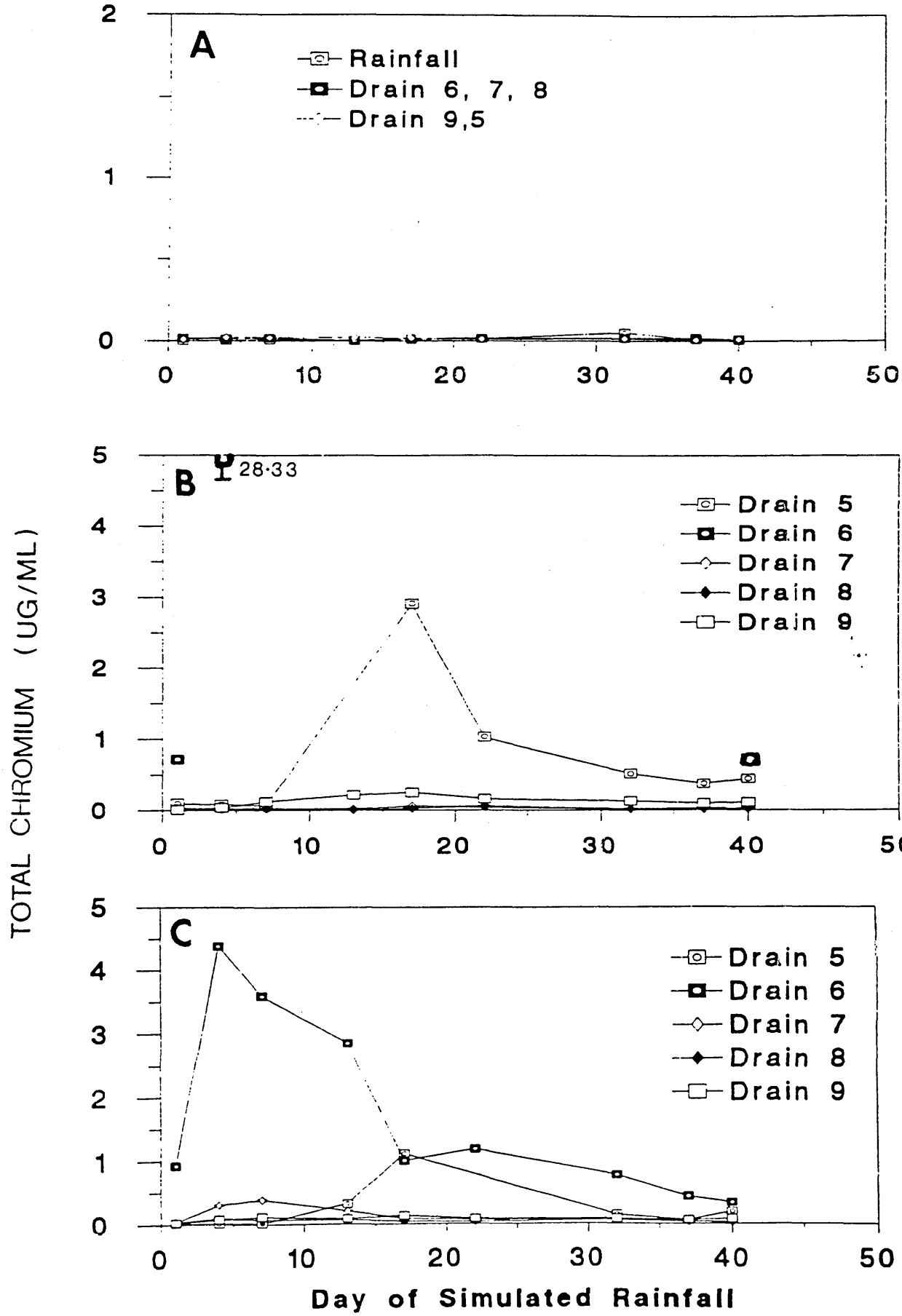


FIGURE.12 Mean Total Chromium concentration (ug/ml) of :-
A - Simulated rainfall and Control soil leachates.
B - Sand amended/treated soil leachates.
C - Treated soil leachates.

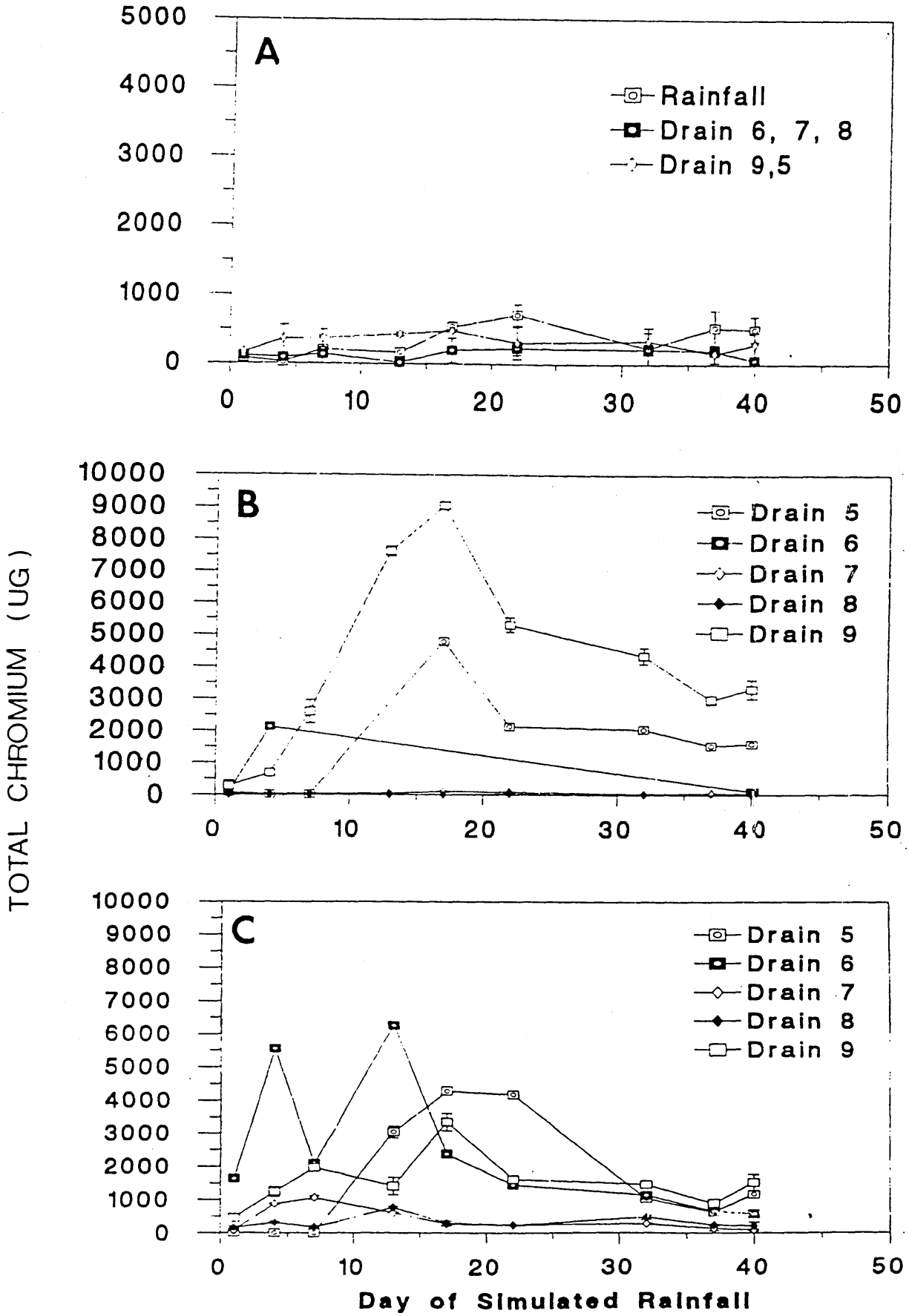


FIGURE.13 Mean bulk Total Chromium content (ug * ml) of :-
A - Simulated rainfall and Control soil leachates.
B - Sand amended/treated soil leachates.
C - Treated soil leachates.

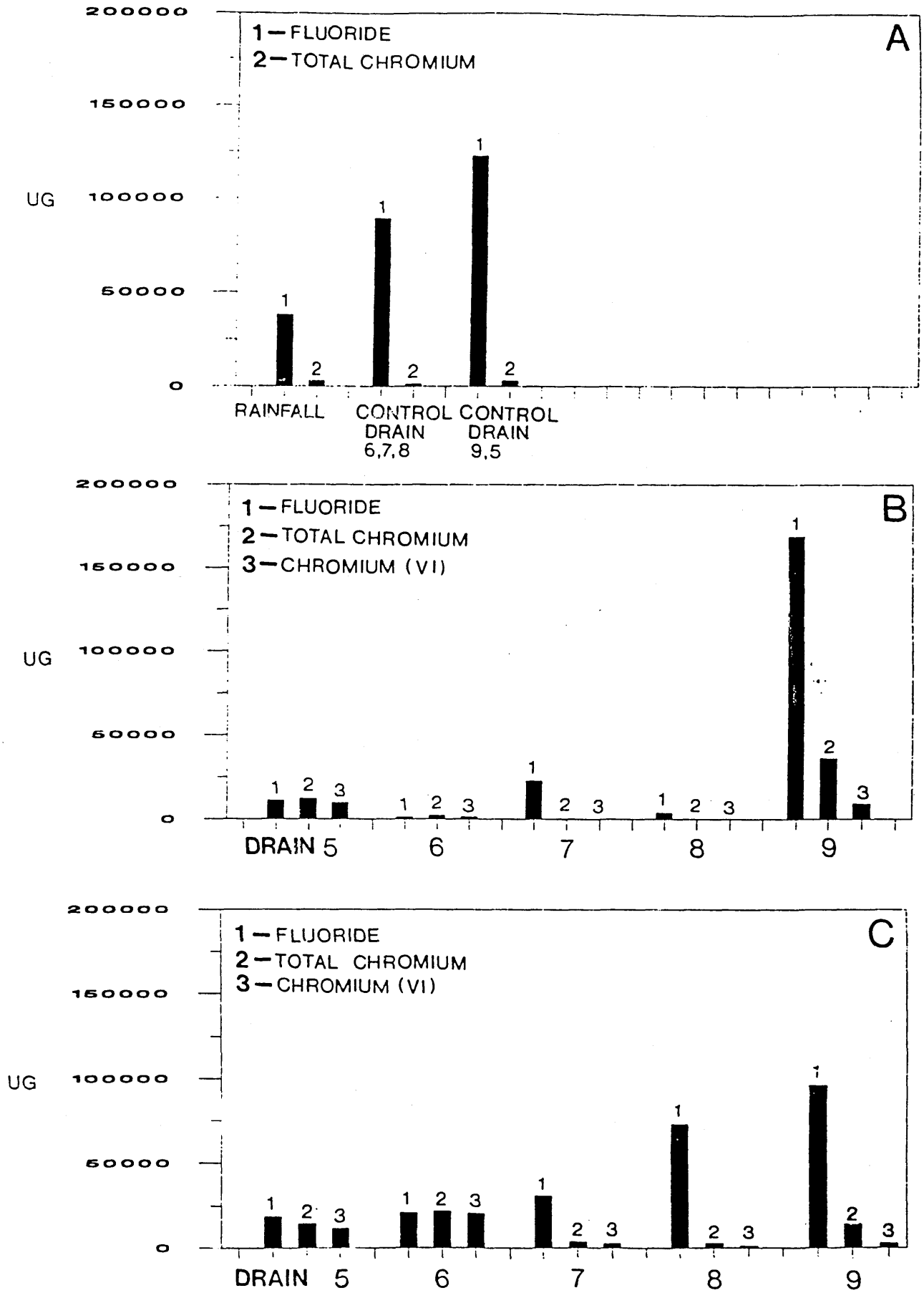


FIGURE.14 Bulk Fluoride, Total Chromium and Chromium (VI) contents over all 'rainfall' applications (ug * ml * 9) for :-
A - Simulated rainfall and Control soil leachates.
B - Sand amended/treated soil leachates.
C - Treated soil leachates.

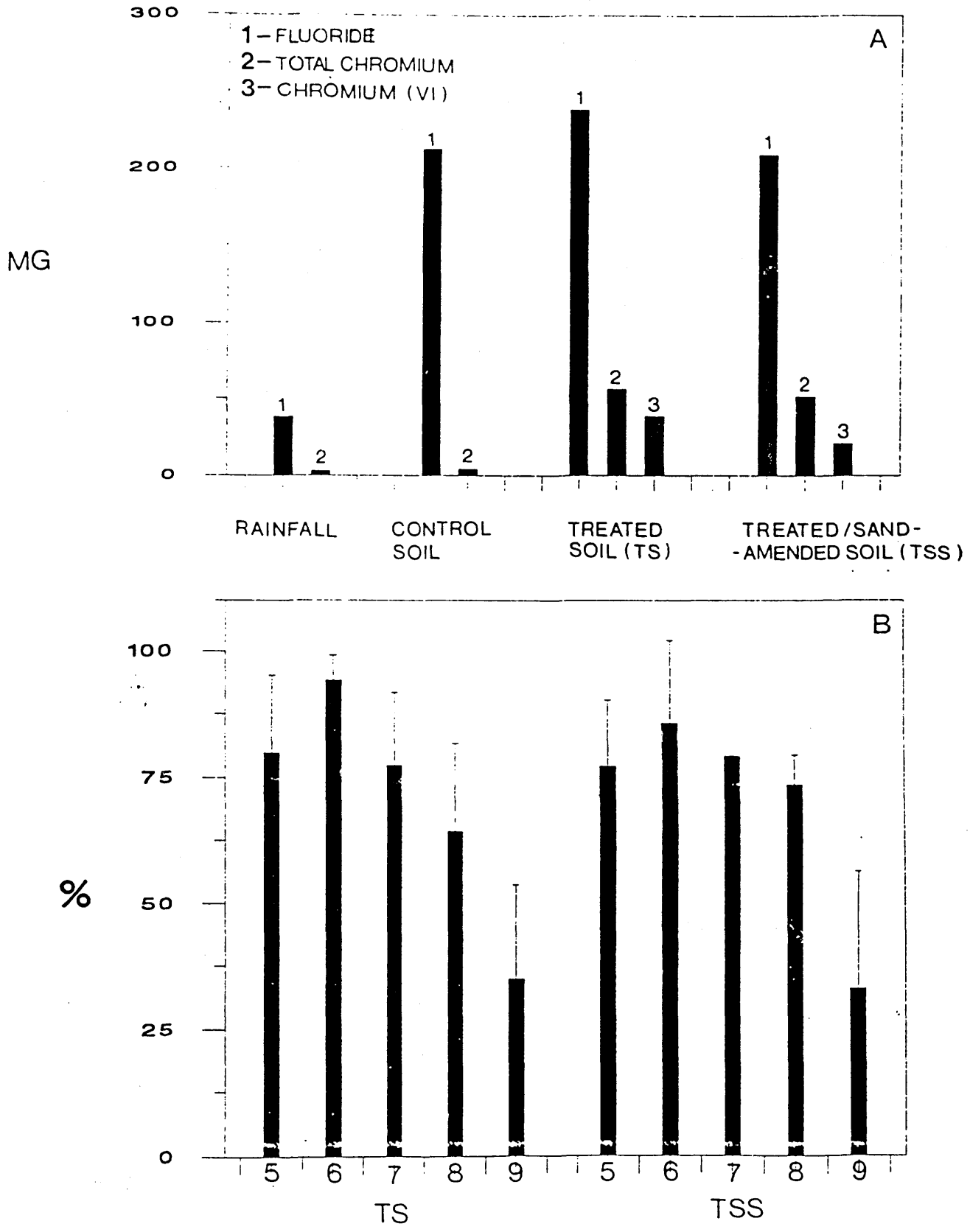


FIGURE.15 A - Bulk Fluoride, Total Chromium and Chromium (VI) contents for all drains over all 'rainfall' applications.

B - Mean Chromium (VI) concentration presented as a percentage of mean Total Chromium concentration of drains 5 - 9 of TS and TSS over all 'rainfall' applications.

Figure 8A and 8C indicates the similar pattern of volume flow through drains of the control soil (for which flow through drains 6, 7, 8 and 9, 5 were combined) and those of the treated soil of the same texture. Figure 8B displays the greatest flow through drains 9 and 5 (see figure 6) due to increased vertical and decreased lateral drainage flow brought about by a lower moisture holding capacity engineered by sand amendment of this treated soil (see figure 7). Drain 6 was prone to blockage for this soil and only 3 samplings were available for analysis.

Figure 9 indicates the great similarity of pH in all the waters throughout the experiment. After an initial fall in pH of leachate from early application of simulated rainfall pH was thereafter maintained in the drainage waters of all the soil profiles at its starting level.

Figure 10 displays the increased concentrations of fluoride (ug/ml) found in the drainage waters of all profiles compared to 'rainfall'. Figure 10B and C shows that the drainage water of the higher profile drains, 6, 7 and 8, of both treated soils retained the highest concentrations of fluoride. However when the bulk loading of fluoride (total ug) in the drainage waters is examined in figure 11, drain 9 of both treated soils (11B and 11C) displays the highest quantity due to the greater volume of flow through this drain shown for both soils in figure 8B and 8C. Figure 11 also indicates that bulk levels of fluoride in the treated soils' drainage waters (11B and 11C) is not that much greater than that of the control soil (11A). There appears also to be an initial flush of fluoride in the drainage water of all soils.

Figure 12 shows the increase in concentration of total chromium (ug/ml) in certain leachates from the treated profiles 12B and 12C compared to the control profile and 'rainfall' at 12A. The treated soils again indicate an early flush of total chromium. Drains 5 and 6 (those closest to the treated pole section) for both treated soils received the greatest concentrations of total chromium (ug/ml). Figure 13B displays the greater bulk loading of total chromium (ug) in the flow through drains 9 and 5 of the sand amended treated soil due to their greater volume (figure 8B). Despite the small volume of flow through drain 6 shown at figure 8C for the unamended treated soil it still initially displays the greatest bulk loading of total chromium (figure 13C) due to its high total chromium concentration (ug/ml) during the early flush. After 13 days, drainage through 9 and 5 shows the greater bulk loading (figure 13C) after the early flush has fallen away.

Figure 14 indicates the bulk loadings of fluoride, total chromium and chromium (VI) multiplied over all simulated rainfall applications. Figure 14A shows that 'rainfall' and control profile drainage water contained no chromium (VI) and that the lower drains (9 and 5) of the control contained greater amounts of fluoride and total chromium. Levels of fluoride in 'rainfall' were substantially less than drainage water levels from any of the profiles. Figure 14B and 14C display the pattern of contaminated drainage water consistent with the different soil textures indicated by drainage flows at figure 8B and 8C. The sand amended soil (14B) shows greatest quantities of fluoride, total chromium and chromium (VI) at drains 9 and 5. The unamended treated soil (14C) displays a more even distribution, with the greatest quantity of total chromium and chromium (VI) appearing in the flow of drains 5 and 6 closest to the treated pole section (figure 6). For both treated soils chromium (VI) makes up the greater part of total chromium concentration in flow through drains other than drain 9.

Figure 15A represents the bulk loadings of fluoride, total chromium and chromium (VI) for all drainage waters over all applications of simulated rainfall. All soil profile drainage waters contained substantially higher levels of fluoride than 'rainfall'. Control profile levels of total chromium were slightly higher than 'rainfall' levels but were insignificant compared to levels in the unamended treated soil (TS) and the sand amended treated soil (TSS). Fluoride levels of control soil drainage were actually higher than those of TSS and only slightly lower than TS. Levels of total chromium in TS drainage water were slightly higher than those of TSS and chromium (VI) made up a greater proportion of TS total chromium than TSS.

Figure 15B indicates the percentage of total chromium which is in the form of chromium (VI) in drainage waters bulked over all 'rainfall' applications. Apart from TSS drainage 7 it shows that the closer the drain to the treated pole section (see figure 6) the greater the percentage of total chromium found as chromium (VI) in the drainage water. Figure 15B also indicates a more even distribution in this percentage across drains 5, 6, 7 and 8 for TSS.

Although no data are presented here, standing leachate samples are being monitored for chromium (VI) content. Initial results indicate that chromium (VI) is being converted to chromium (III) on standing possibly due to the oxidation of the soluble organic matter content of the leachate samples.

6. DISCUSSION.

The results presented above show that the model system described here allows an accurate assessment of the amounts of preservative chemicals lost from treated timber to adjacent soil water under conditions of heavy simulated rainfall. Thus this system has the advantage over field studies that any possible contamination of both soil and groundwater can be assessed under controlled conditions.

Figure 15A shows the similarity of preservative contamination of leachate samples in the 2 models containing preservative treated timber and indicates reasonable reproducibility of the model system.

This figure also shows that the portion of the fluoride content of soil water due to the preservative treatment is a small fraction of that of the normal soil water fluoride content as shown in the control model (15A).

The total chromium content of leachates due to losses from the preservative treated timber is substantially higher than control soil levels (figure 15A). However the total quantity of chromium accumulated in leachates of each treated pole model over the complete experimental period amounted to approximately 50 mg.

The results presented in this paper will be examined in conjunction with the associated soil and plant analysis data to provide an overall description of the environmental impact of the preservative treated poles and will be published in due course.

ACKNOWLEDGEMENTS.

The authors acknowledge the financial and practical assistance given by Cobra Wood treatment (UK) and Rentokil Ltd. The poles were kindly provided by Hydro Electric Plc. Thanks are also due to the Scottish Crop Research Institute, Mr Ian Foot of Twyford Seeds Ltd and Margot Dunnachie for drawings of the model system.

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Section 1 Biology

Assessment of Dehydrogenase Activity, Fluoride Content and Total Chromium Content of Soil Profiles Exposed to Preservative Treated Wood within a Model System.

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ASSESSMENT OF DEHYDROGENASE ACTIVITY, FLUORIDE CONTENT AND TOTAL CHROMIUM CONTENT OF SOIL PROFILES EXPOSED TO PRESERVATIVE TREATED WOOD WITHIN A MODEL SYSTEM

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ABSTRACT

The development and prospective use of a closed model system to facilitate study of a number of indicators of environmental impact of wood preservatives under laboratory conditions has been described (IRG/WP/2395-92). Chemical analysis of leachate samples collected from drained soil profiles containing creosoted pole sections remedially treated with a chromated fluoride preservative indicated small increases in fluoride and chromium concentrations. This paper details measurement of dehydrogenase activity and chemical analysis of soil samples recovered from the surface layers of the model soil profiles adjacent to treated pole sections. Reduced levels of dehydrogenase activity were associated with increased soil concentrations of leached preservative components and lower organic matter content. Findings are discussed as part of an assessment of environmental impact of the remedial treatment in the field.

Keywords: Creosoted poles; Rentex; model system; fluoride; chromium; dehydrogenase activity; environmental impact.

INTRODUCTION

An effective model system for assessing the environmental impact of preservative treated timber in soil contact should allow measurement of both chemical and biological parameters in the soil environment adjacent to the treated timber. Thus not only the presence of preservative components in both soil and ground water should be monitored but also any effects these may have on the soil flora and fauna where practicable.

A model system has recently been described (Sinclair et al 1992) and the first results obtained from it (Sinclair et al 1993) describing movement of fluoride and chromium in soil leachate waters. The model systems contained creosote treated pole sections remedially treated at the ground-line region with Rentex. Fluoride based remedial formulations for poles have been used for many years (Steinherz, 1939; Becker, 1973). Rentex, a more recent formulation, is a paste containing sodium fluoride, ammonium bifluoride and sodium dichromate.

The preservative is applied by a series of injections to a zone of the pole between 35 cm above and 35 cm below the ground line. A bitumen coat is painted over the treated zone and an aluminium sheath placed round the part of the pole above the ground line (Morris and Calver, 1987).

Previous work (Bruce and King,1989) showed Rentex to be an efficient remedial preservative. However chemical analysis of soil samples taken next to treated on-line poles at a number of sites indicated leaching of chemical components fluoride and chromium (Sinclair et al, 1992).

In view of these findings closed model systems containing two aged remedially treated creosoted pole sections and one control creosoted pole section were constructed to allow assessment under laboratory conditions of any leaching of preservative components to their adjacent soils and of any accompanying environmental effects (Sinclair et al, 1992, 1993).

Chemical analysis of leachates draining through the model systems showed small but significant increases of fluoride and chromium in the drainage waters of the two Rentex treated models compared with the control (Sinclair al, 1993). It was also observed that higher concentrations of fluoride were found where the total volume of leach water was low and close to the pole section. However when the total amounts of fluoride from the Rentex treated and untreated models were compared it was found that the overall difference was small compared with background levels of fluoride which appears as a natural constituent of the soil used.

The total chromium content of the leach waters (average total volume 290 litres) from the Rentex treated models, some 50 mg, was much higher than that of the control model which amounted to 4 mg.

These amounts of chemical components in the leach waters suggest that there is no serious threat of ground water contamination by leaching from the treated poles in the field as most of the fluoride and chromium lost from the poles (Sinclair et al,1993) appears to be retained by the soil.

To assess the effects of pesticides on soil microflora the United States Environmental Protection Agency suggested measurement of dehydrogenase or phosphatase activities (US EPA ,1978). Burns (1978) considered methods and interpretion of results to be inadequate. Davies and Greaves (1981) concluded that the wide range of soils used with greatly differing enzymic activities, and the poorly understood relationship between soil fertility and the activities of enzymes secreted by soil micro-organisms has frequently been responsible for contradictory results.

Nonetheless Forstner (1988) argued that the most relevant mechanism of toxicity of heavy metals in soil is the chemical inactivation of enzymes. As part of an assessment of the environmental impact of remedially treated poles in soil, it is therefore appropriate to examine dehydrogenase activities of soils with raised chromium levels as found adjacent to the Rentex treated poles in the field.

MATERIALS AND METHODS

MODEL SYSTEMS

Details of the three model systems used in this experiment have been previously described (Sinclair et al, 1992 and 1993). The control model (CS) contained an aged 2m creosoted pole section in a sandy loam texture soil. One model (TS) contained a Rentex treated creosoted pole in the same type of soil as the control model. The third model (TSS) contained a similar Rentex treated pole in a soil amended with one part of washed sand to two parts of soil.

The soils in the three models were maintained at field water capacity for a total of 175 days. For 40 days during this period the pole sections and models were subjected to a regime of simulated rainfall by application of tap water via a calibrated mist/sprinkler system. The total applied rainfall amounted to the equivalent of half the annual rainfall at the experimental field site (Sinclair et al 1993).

SOIL SAMPLES

Soil samples (5 cm x 5 cm x 15 cm depth) were removed from numbered positions within each model (fig. 1) and were bulked and homogenised as shown by the numbers in figure 1 giving 6 samples for models TS and TSS. Fewer control model samples were used. Positions 1 and 3 were combined and positions 5 and 6 only used as marked in fig. 1 i.e. a total of three control samples.

The moist soil samples were individually homogenised and seived through a 2 mm stainless steel seive. The samples were then placed in unsealed plastic bags within plastic boxes containing a 3 cm deep layer of water. The boxes were loosely sealed and left for one week at 20 deg C at a relative humidity of 75% to allow them to reach equilibrium moisture content. They were then sub-sampled to determine moisture, fluoride and total chromium contents.

A 200 g amount of each of the 15 soil samples was adjusted to 20 % w/w moisture content. These were split into two 100 g portions. One of these portions was supplemented by mixing with 1 g of milled Rye meal previously held at 120 deg C for 24 hours. Each of the two types of 100 g sample was split into 4 sub-samples of 25 g. Each sub-sample was placed into a loosely stoppered 70 ml glass sample bottle to a depth of about 2 cm. This gave 120 such sub- samples in total.

The bottles were stored in a covered ventilated plastic tray, containing a 3 cm deep water layer, at 18 deg C and relative humidity 85 %, for 4 weeks.

At intervals of 18 hours, (represented graphically as week 0), 1, 3 and 4 weeks after supplementation of the soils, 15 pairs of bottles, one supplemented and the other not, were removed and tested for dehydrogenase activity.

DEHYDROGENASE ACTIVITY

Dehydrogenase activity was measured by the method of Casida et al (1964) as modified by Mowe (1983).

Four 1.5 g sub-samples were removed from each bottle, three for dehydrogenase activity measurement and one for dry weight correction. The three replicates were each weighed into a screwtop test-tube (120mm x 15mm) containing 15 mg calcium carbonate and 2 ml of 0.75% w/v 2,3,5,-triphenyltetrazolium chloride (TTC) solution. After thoroughly mixing the contents of the tubes on a vortex shaker, they were sealed and incubated at 30 deg C for 24 hours in the dark.

After incubation ethanol (5 ml) was added to each tube and mixed 5 minutes on a vortex shaker. On settling, the supernatant liquid in each tube was decanted into a centrifuge tube and the remaining soil particles were rinsed with ethanol (3 ml) and decanted again. The total decanted liquid was centrifuged (x 4000 g) for 5 minutes to separate remaining soil particles. A portion of the clear liquid was transferred to a glass cuvette and the absorbance of the resultant 2,3,5, triphenyltetrazolium formazan (TTF) read at 485nm. The concentration of TTF for each sample was determined by reference to a prepared standard calibration graph of TTF in ethanol.

The Rye meal used for supplementation showed no dehydrogenase activity and each set of analysed soil samples was accompanied by three reagent blanks. Dehydrogenase activity is expressed in units of μ mols TTF/ g min.

FLUORIDE AND CHROMIUM ANALYSIS

Soil sub-samples were finely ground and taken into solution following alkali fusion using a method described by McQuaker and Gurney (1977) for fluoride analysis of soils and modified by Sinclair et al (1991) to include chromium. Fluoride was determined by selective ion electrode and chromium by atomic absorption spectrophotometry. Fluoride and total chromium results are expressed as μg element/ g dry soil.

RESULTS

Dehydrogenase and preservative component concentrations are presented in figures 2 to 6.

Figures 2 and 3 show as expected that supplementation of the soil samples with Rye meal significantly increases mean dehydrogenase activity. The supplemented soil samples (fig. 3) show an initial increase from weeks 0 to 1 decreasing then at weeks 3 and 4. This trend is not obvious in the unsupplemented samples (fig. 2).

Supplemented samples from the control model showed significantly greater mean dehydrogenase activity at position 1-3(next to the pole)than at position 6 (at weeks 1 and 3) and position 5 (at weeks 3 and 4), with $P \leq 0.050$. Unsupplemented soils from this model only showed significantly different dehydrogenase activity at 4 weeks, with sample 1 greater than sample 6, for $P = 0.050$.

For model TS soil samples at positions 3 and 1 consistently displayed reduced mean dehydrogenase activities compared with the other sample positions for both supplemented and unsupplemented soils.

With unsupplemented TS soils the mean dehydrogenase activity of sample 3 (next to the pole) was significantly lower than all others at week 1 ($P = 0.001$) and lower than samples 2,4,5,and 6 at 0,3, and 4 weeks ($P \leq 0.003$). Soil sample 1 showed a lower mean dehydrogenase activity than samples 4,5, and 6 at week 3 and 2,4,5,and 6 at week 4 ($P \leq 0.0005$).

Supplementation of TS soil samples enhanced these differences (Fig. 3). Soil sample 3 is thus significantly lower than all others at every sampling time period except for position 1 of week 3.

For model TSS the mean dehydrogenase activity in the soil samples showed a similar pattern to that of model TS. Soil samples from positions 1 and 3 were again consistently lower in activity than the others but to a lesser degree than in the non-sand amended soil.

Comparing the unsupplemented soil samples of the three models in general (fig. 2) it can be seen that the sand amended soil in model TSS has consistently lower activity than the soils in the other two models with the exception of the samples at position 3 where models TSS and TS become about equal at weeks 0,1 and 4.

Likewise comparison of the three models using the supplemented soil samples (fig.3) showed the sand amended soil model to have the lowest activity overall with both the treated pole models again displaying the same pattern of activity across the six sampling positions.

The mean dehydrogenase activity at all sample positions within each individual model displayed at each sampling time (fig.4) again shows that supplementation increases the soil dehydrogenase activity but does not significantly alter the general trend for the control and unamended soils. It does however bring the sand amended soil model's general pattern more into line with that of the control and unsupplemented soil models.

The combined means of dehydrogenase activity for each sample position over the four sampling times (fig.5) also show that supplementation increased activity without altering the general trend across the sample positions.

The mean fluoride and mean total chromium contents of soil samples taken for each model at each dehydrogenase sampling position are shown in figure 6. Total chromium refers to the total amount of the element in all its oxidation states. Chromium is applied to the poles as Cr(VI) in sodium dichromate but this highly oxidising form gradually changes to Cr(III) in wood and soil. The analysis method employed here determines the total chromium content of the soil samples.

Soil samples from the control model (CS) showed no significant differences with sampling position for either fluoride or chromium.

For the unamended soil model (TS) sampling positions 1 and 3 next to the pole had statistically greater concentrations of both fluoride and chromium than those at positions 2,4,5 and 6, ($P = 0.001$) within the same soil profile.

For the amended soil model (TSS) there were no significant differences in chromium content amongst the sampling positions but the fluoride concentrations at positions 1 and 3 next to the pole were significantly greater than those further away ($P \leq 0.0005$).

A comparison of the fluoride and chromium concentrations at corresponding sample positions away from the pole for the control and sand amended models clearly shows the effect of the sand amendment. The control model shows significantly higher levels of both elements at positions 5 and 6.

The unamended model (TS) had significantly greater concentrations of fluoride and chromium at positions close to the pole compared with the control model ($P = 0.004$).

DISCUSSION

Dehydrogenase activity was found for all soil samples tested in each of the three model systems used regardless of the presence of the remedially treated poles in two of the models.

It has been observed previously (Green 1988) in small block burial experiments that some soil samples adjacent to Sitka Spruce sapwood blocks treated with 5% CCA showed no dehydrogenase activity after a period of burial. Both small block burial tests (Green, 1988) and larger scale pole sections used in fungal cellar studies (Hainey, 1992) have shown stimulation of dehydrogenase activity due to the presence of the wood followed by decreased activity as the preservative components leach from the treated wood blocks and sections.

The control model system in fact containing an aged creosoted distribution pole showed an increased level of microbial activity close to this pole compared with samples taken at a distance from it.

The major factor influencing dehydrogenase activity in this experiment was the amendment of the soil using sand, which resulted in considerably decreased dehydrogenase levels in the sand amended model. This reduction is probably associated with the reduced organic matter content of the amended soil. Sand amendment produces improved water drainage and causes changes in the patterns of soluble preservative components' concentrations in leach waters moving through the models (Sinclair et al 1993). Efficient drainage is an important factor in the growth of both plants and micro-organisms in model systems of this type.

Supplementation of the organic matter content by addition of sterile Rye meal did not mask the decrease in dehydrogenase activity due to sand amendment. However the use of supplementation highlighted comparisons amongst the other factors influencing dehydrogenase activity.

Both models containing the remedially treated poles showed significantly reduced dehydrogenase activity at positions where increased levels of the preservative components fluoride and chromium were found in the soil i.e. next to the treated timber in contrast to the increased activity found at this position for the control pole. However although a statistically significant reduction in soil dehydrogenase activity due to the Rentex treatment has been found this effect is restricted to a small soil section immediately adjacent to the treated timber and would not be expected to reduce soil fertility outwith this region.

Dehydrogenase activity has thus been shown to serve as a sensitive indicator of the influence of toxic wood preservative components leached from treated timber into adjacent soil in small block burial tests (Green, 1988), fungal cellar tests (Hainey, 1992) and model systems (Sinclair et al, 1992,1993) as used for the current work.

The use of these model systems has therefore shown that there is very little deleterious effects on soil fertility and ground water arising from timber poles remedially treated with Rentex.

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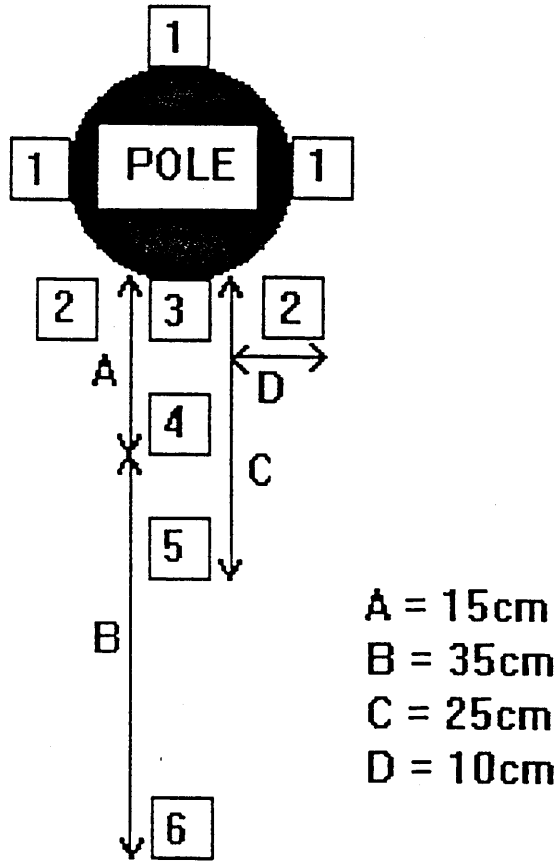

FIGURE 1. Sampling plan for removal of soil cores from model profiles.

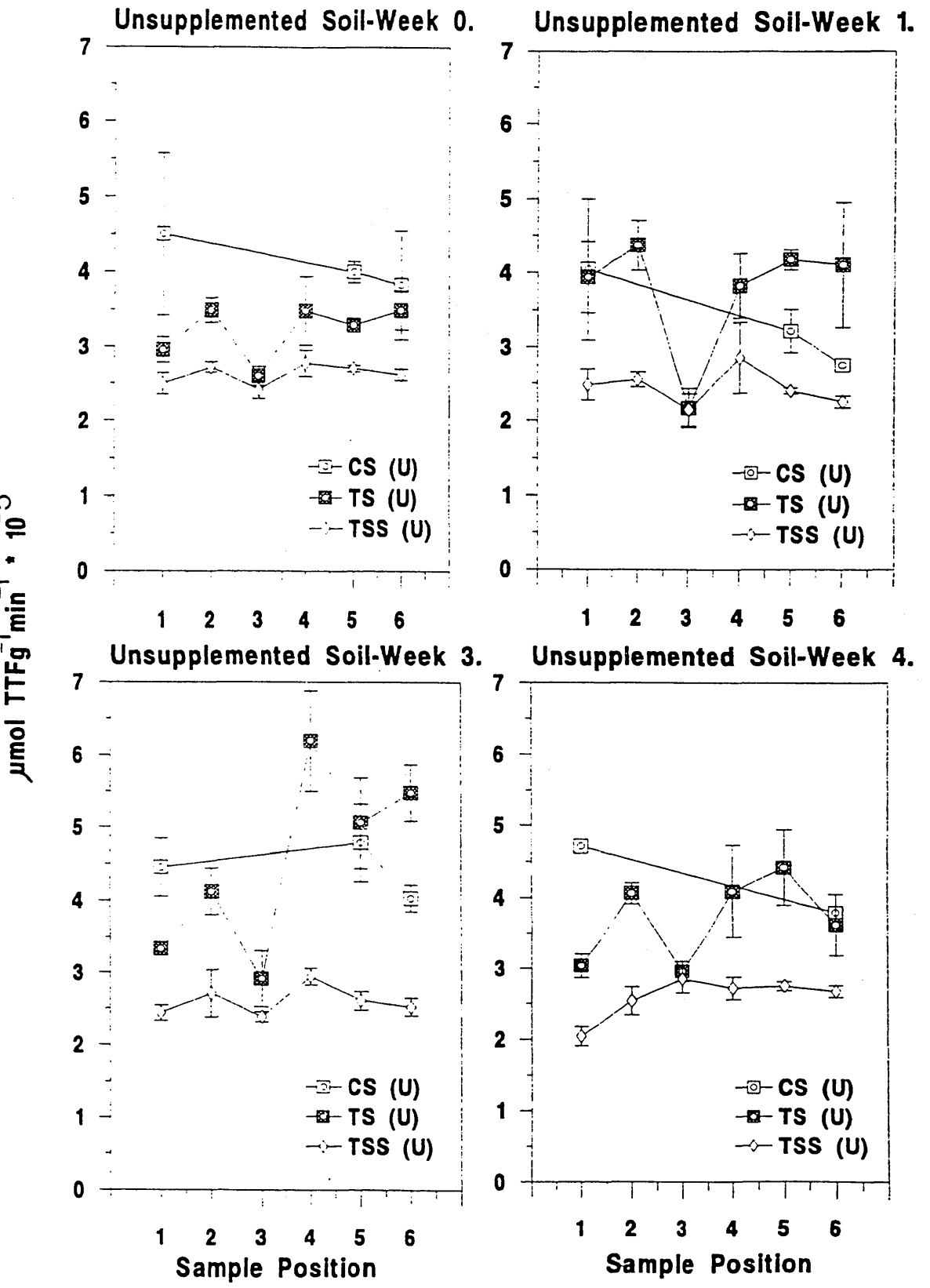

FIGURE 2. Mean dehydrogenase activity of unsupplemented soil samples from CS, TS and TSS soil profiles at 0 1, 3 and 4 weeks (St. dev. for means of 3).

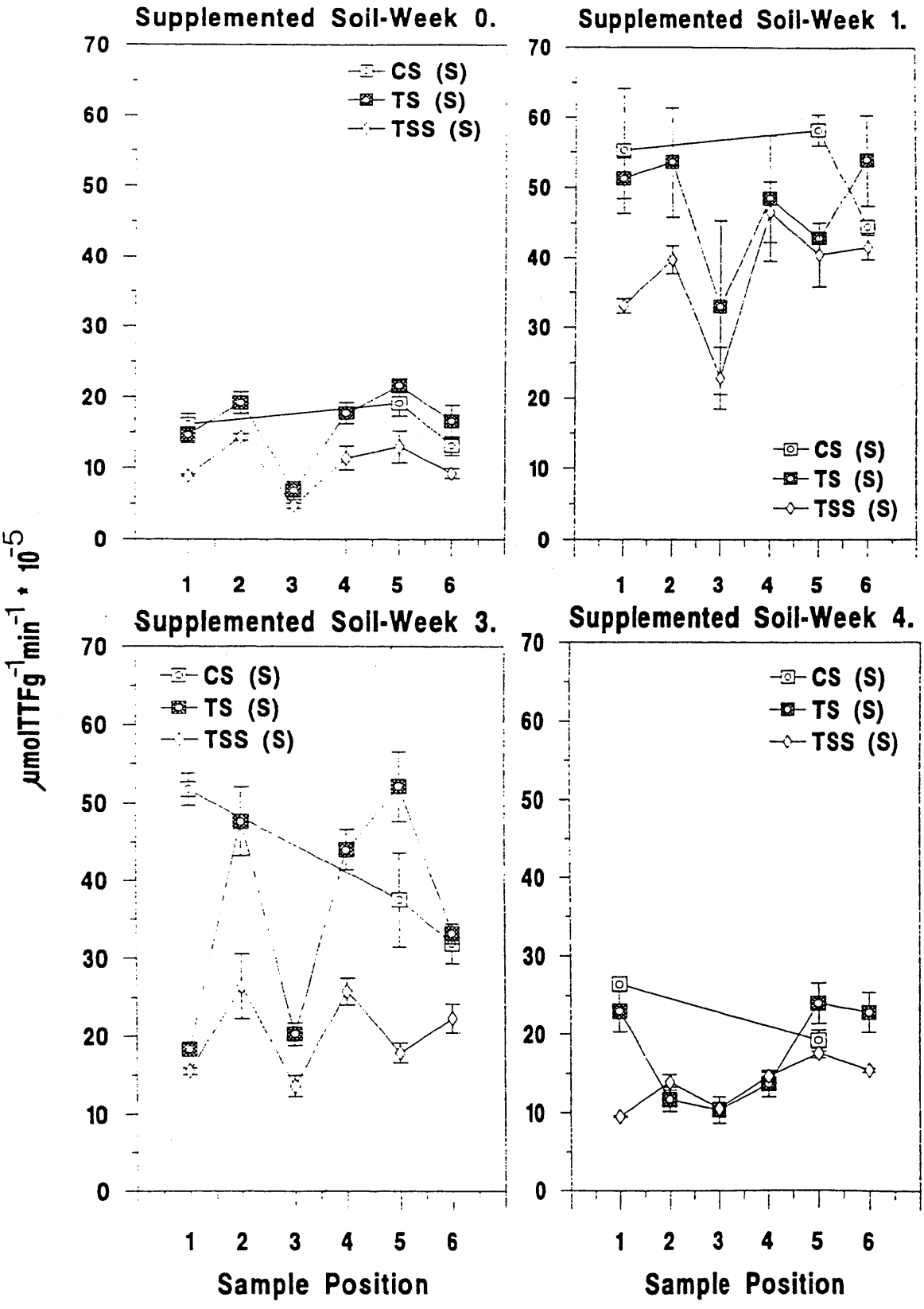

FIGURE 3. Mean dehydrogenase activity of supplemented soil samples from CS, TS and TSS soil profiles at 0, 1, 3 and 4 weeks (St. dev. for means of 3).

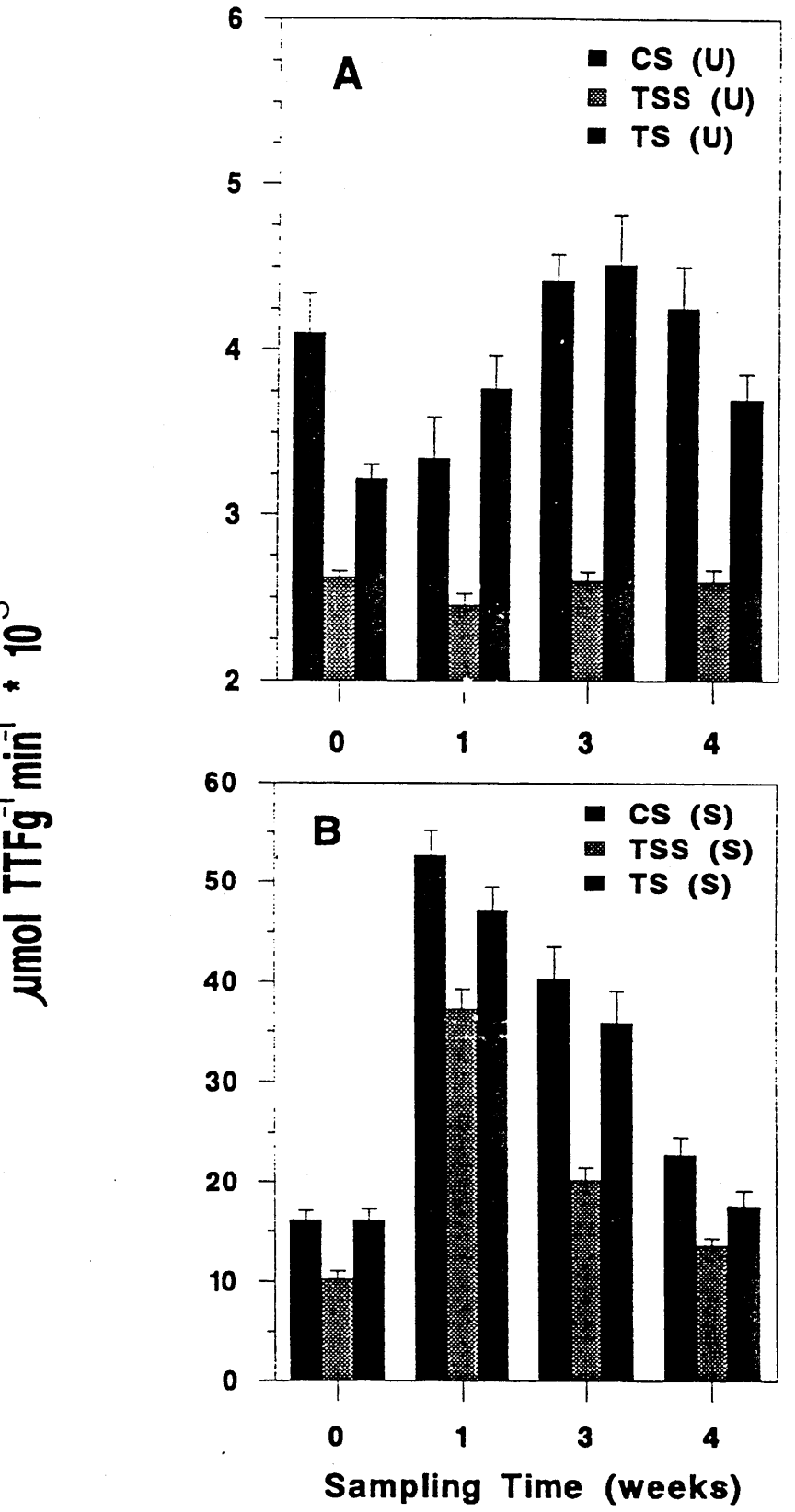

FIGURE 4. Mean dehydrogenase activity of soil samples (A/Unsupplemented and B/Supplemented) from CS, TS and TSS soil profiles combined for 0, 1, 3 and 4 weeks (St. errors for means of 9/CS and 18/TS and TSS).

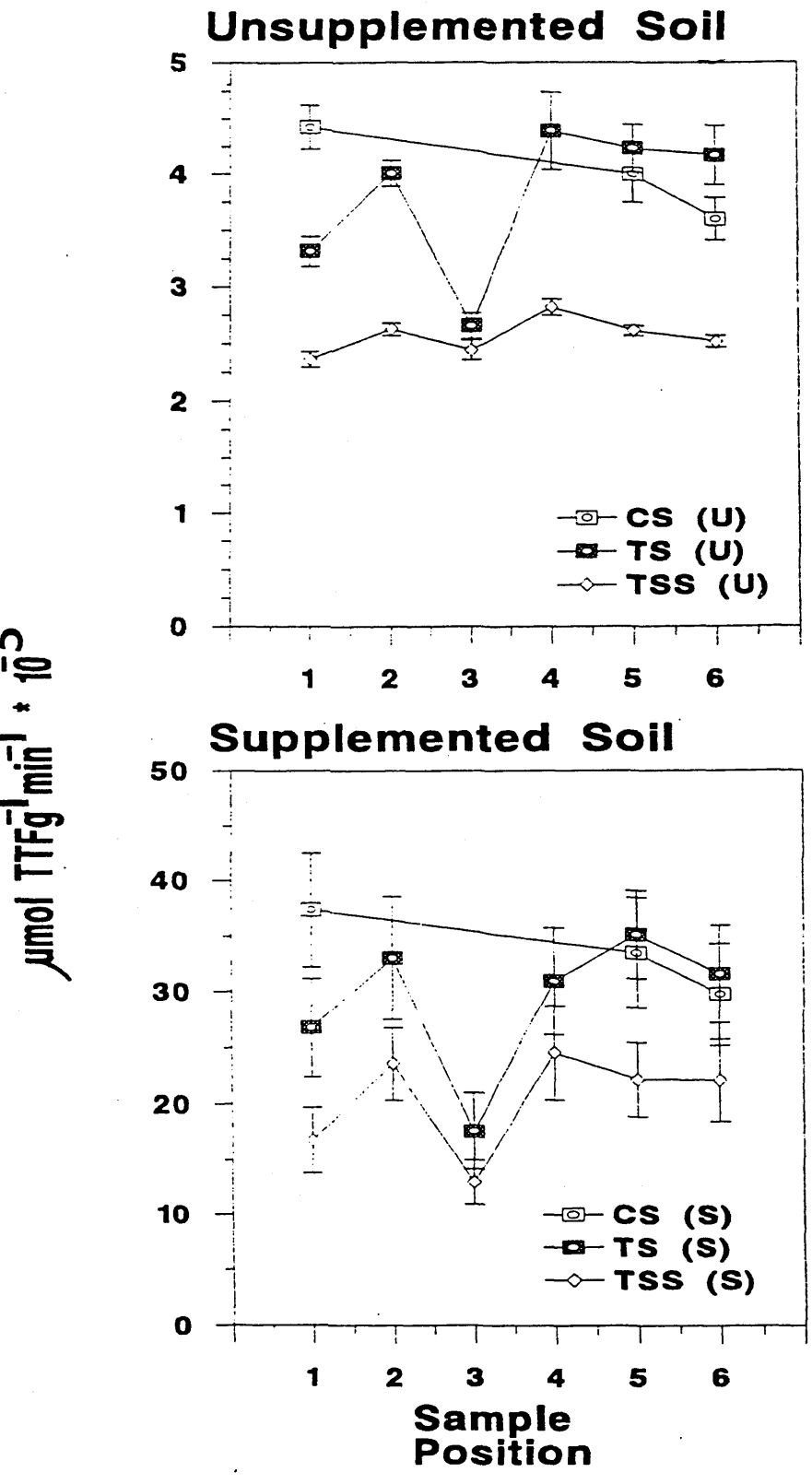

FIGURE 5. Mean dehydrogenase activity of unsupplemented and supplemented soil samples at 0, 1, 3 and 4 weeks (from CS, TS and TSS soil profiles) combined for each sample position (St. errors for means of 12).

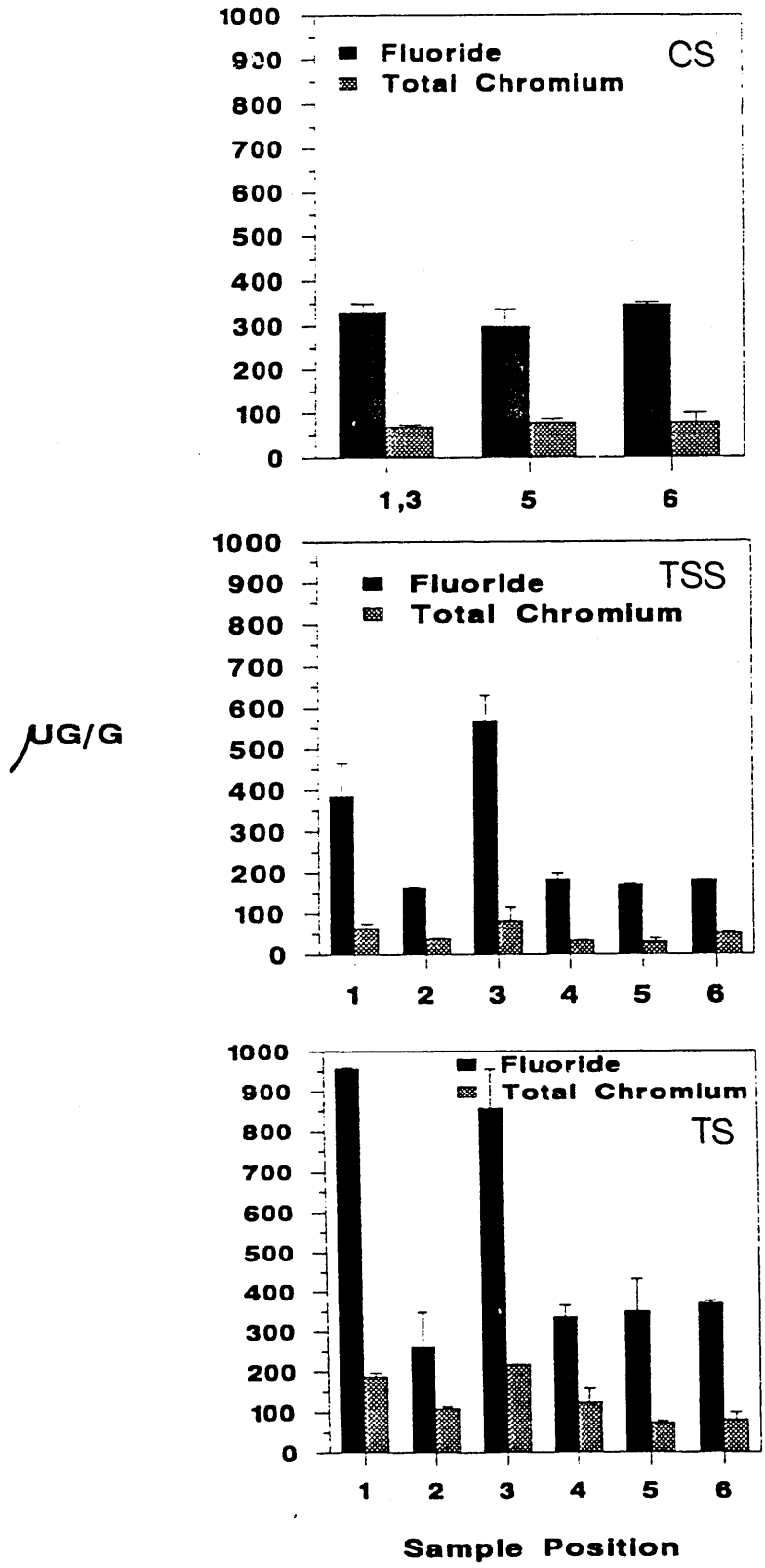

FIGURE 6. Mean fluoride and total chromium contents of soil samples from the upper 15cm of CS, TSS and TS soil profiles (St.dev for means of 2).

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THE INTERNATIONAL RESEARCH GROUP ON WOOD PRESERVATION

Section 3 Wood Protecting Chemicals

Assessment of the Effects of Rentex Remedial Treatment on Some Wood Pole Inhabitant Micro-organisms

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ASSESSMENT OF THE EFFECTS OF RENTEX REMEDIAL TREATMENT ON SOME WOOD POLE INHABITANT MICRO-ORGANISMS.

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ABSTRACT

The effects of a remedial ground-line treatment using Rentex, a stabilised paste containing a mixture of fluoride and dichromate salts, on the micro-organisms inhabiting a sample of some 160 creosoted, on-line, electricity distribution poles, have been investigated as part of an appraisal of the efficacy of this remedial treatment. Half of the poles were treated and the other half used as controls alternately along the line. Wood cores were removed from these poles immediately before treatment and at a series of 4 time periods for up to 16 months after treatment. The uncreosoted sections of these cores adjacent to the heartwood were incubated on nutrient agar plates to identify the presence of basidiomycetes, bacteria and mould organisms. The low number of isolations of basidiomycetes in particular, the main decay fungus in creosoted poles *Neolentinus lepideus*, precludes meaningful conclusions with respect to these organisms. However comparisons between the control and treated poles with respect to numbers of micro-organism free poles indicates a treatment effect superimposed on a seasonal population effect. Over the 16 month period of study the remedial treatment produced a consistent significant reduction in numbers of isolated organisms. This result confirms expectations from studies of the distribution of preservative elements in poles remedially treated with Rentex and the toxicity of this formulation against moulds.

1. Introduction

Internal decay of creosoted distribution poles occurs when the creosote treatment has failed to penetrate completely the susceptible sapwood, due to inadequate treatment at the plant or because the wood was not properly seasoned beforehand (Smith and Cockcroft, 1967 a). When decay has become established in untreated sapwood, it is not uncommon for it to spread into the more resistant heartwood (Smith and Cockcroft, 1967 b) which is unprotected by creosote pre-treatment. Internal decay predominates at the groundline (Chambers, 1963; Smith and Cockcroft, 1967 b, c; Anon, 1971; Becker, 1976) where moisture and oxygen

conditions combine to provide an environment conducive to the growth of decay fungi, including *Neolentinus lepideus Fr.*, the basidiomycete most commonly associated with this type of decay in the United Kingdom (Cartwright and Findlay, 1958; Bruce, 1983; Smith and Cockcroft, 1967 c).

Consequently, the electricity supply industry has for many years attempted to control internal decay of distribution poles by the use of waterborne fluoride preservatives as a groundline remedial treatment (Steinherz, 1939; Chambers, 1963; Becker, 1973, 1976). Injection of the original DFA or "Cobra" salts containing dinitrophenol, sodium fluoride and arsenic (III) oxide, was discontinued in 1986 in the United Kingdom owing to concerns over health and safety regarding the arsenic and dinitrophenol components. Rentex, a stabilised paste containing sodium fluoride and ammonium bifluoride as fungicides and sodium dichromate as a "fixative", is a more recent preservative formulation investigated (Bruce and King, 1989; Sinclair et al, 1991) as a remedial treatment.

Based on field studies of the DFA salts and a survey of toxic limit tests (Smith and Cockcroft, 1967 b, c), Smith and Cockcroft (1967 c) expected that fluoride concentrations in poles would give protection against internal decay by *N. lepideus* for approximately 8 - 9 years. The findings of Sinclair et al (1991), from a field study of pole sections up to 12 months after Rentex treatment, in conjunction with unpublished toxic limit data from this laboratory, indicate that fluorides diffused from preservative injection sites to provide fluoride concentrations within uncreosoted sapwood and heartwood which were generally above the toxic threshold for *N. lepideus*. Statistical analysis of this data (Sinclair et al, 1991) together with data from a continuation of the study (unpublished), indicated that fluoride diffusion was enhanced in timber of higher moisture content, which has been shown for fluorides in a number of studies (Liese and Schubert, 1941; Buro and Becker, 1956; Becker, 1959).

These indications of the efficacy of Rentex remedial treatment are supported by a previous field evaluation (Bruce and King, 1989) in which *N. lepideus* inocula in wood poles were eliminated by remedial treatment with a Rentex formulation. However, the specific relevance of this latter work to the studies of Sinclair et al (1991) is in doubt due to observations, on pole sectioning (unpublished), that more preservative injections were made to the timber studied by Bruce and King (1989).

Therefore, in order to support these indications of the efficacy of the Rentex remedial treatment (Sinclair et al., 1991), a field study was also undertaken , to determine the effects of remedial treatment on naturally occurring populations of

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micro-organisms, including *N. lepideus*, colonising internal uncreosoted groundline areas of creosoted distribution poles.

2. MATERIALS AND METHODS.

2.1. Field operations.

2.1.1. Excavation and measurement of distribution poles.

In December 1989, 240 creosoted distribution poles, in service since 1958 and situated at Glen Clova in the east of Scotland, were excavated to a depth of 0.5-0.75 m and pole circumferences recorded at the groundline. The height of each pole above the groundline and its depth below it was recorded from the data stamp present on each pole surface. In this study 160 of these poles were used.

2.1.2. Pre-Treatment core samples.

Four wood cores of bore 0.5 cm and measuring approximately half the pole diameter in length were recovered from each of the 240 poles using a Mattson auger, dipped in industrial alcohol and flame sterilised between each sample. Cores were removed from positions, in a vertical line, at 35 cm and 17.5 cm above the groundline, 17.5 cm below the groundline and at the groundline itself with the augur positioned at right angles to the pole surface. Each core was immediately inserted aseptically into a labelled sterile screw top test tube to be used for later isolation and identification of inhabitant micro-organisms (section 2.2.1.) and for creosote depth measurements in the laboratory.

2.1.3. Measurement of pole moisture contents.

The % moisture content of each pole was recorded using a 'Protimeter' conductivity moisture meter attached to an elongated probe developed by the Midlands Electricity Board. The probe was fully inserted into the groundline borehole and pressed firmly to the pole interior thereby measuring conductivity across 2 contact points on the end of the probe and hence % moisture content. The core sample boreholes were plugged with softwood dowels, which, for poles awaiting preservative treatment, had been pressure impregnated with a 5 % solution of Rentex.

2.1.4. Pole treatment.

After core removal and measurement of pole moisture content every alternate distribution pole of the 160 excavated was treated with Rentex, a mixture of sodium fluoride, ammonium bifluoride and sodium dichromate with fillers (Sinclair et al, 1991). Application was carried out by forcing preservative paste into the pole to a depth of 6.5 cm via a hollow injection needle propelled by a mechanical pump. Pole diameter determined the number of injections, up to 120, applied in a defined diamond pattern to the pole in a treatment zone which extended approximately 35 cm above and below the groundline. A bitumen coating was applied as waterproofing to the entire treated area and an aluminium sheath was fixed around the treated region above the groundline. This provided 120 treated and 120 untreated poles, each treated pole flanked by 2 untreated control poles and vice versa. Treated and control poles were therefore equally represented at any particular site within the larger Glen Clova area. Preservative injections ensured that the sealed boreholes were positioned centrally within a diamond pattern of 4 injections characteristic of the treatment method. After preservative treatment or sealing of boreholes in the case of untreated poles, soil excavations adjacent to poles were refilled.

2.1.5. Pole ranking system.

Eighty poles from each group of 120 were selected as follows according to their moisture contents. Each group of 120 remedially treated and 120 control poles were ranked from lowest to highest moisture contents based on the 1st recorded pole moisture contents as this pole parameter was likely to have a major influence on preservative efficacy.. Each group (treated and control) was split into 20 smaller groups of 6 (numbered 1-6) starting with poles of lowest moisture content and proceeding through to poles of highest moisture content. This grouping system provided 20 poles numbered 1, 20 numbered 2 and so on through to 20 poles numbered 6, for each group of 120 control and treated poles. This ranking system provided, in every group of 20 treated and 20 control poles numbered 1 or 2 or 3 etc., 10 poles of lower and 10 of higher moisture content for each of the 4 time periods used (section 2.1.6.).

2.1.6. After-Treatment core samples.

At 1 month after remedial treatment 20 treated and 20 control poles, each numbered 1 according to the ranking system, were excavated. The aluminium sheath fitted to each treated pole was carefully removed. Three core samples were removed as before, from each pole next to the original 3 lower sampling positions (section

2.1.2.) from the middle of the adjacent diamond pattern of injections for isolation of micro-organisms (section 2.2.1.). Pole moisture contents were recorded at the groundline to confirm the original moisture status of each pole and core boreholes were plugged as before (section 2.1.3.). Aluminium sheaths were refitted to treated poles and excavations refilled.

At 3, 6 and 16 months after remedial treatment a separate group of 20 treated and 20 control poles, each numbered 2, 3, and 4 respectively were similarly excavated and sampled. Thus, a total of 80 treated and 80 control poles were core sampled a 2nd time and core samples from treated and control poles of high and low moisture content were equally spread over all core sample collections.

2.2. Laboratory operations.

2.2.1. Core samples for isolation and identification of micro-organisms.

The core recovered from 35 cm above the groundline of each pole section at the 1st sampling (section 2.1.2.) was used to determine creosote depth as a percentage of pole radius, calculated from the recorded circumference of each pole .Examination of these cores indicated that no uncreosoted sapwood was present in any pole.

The creosoted portion of the other three cores recovered from each pole (sections 2.1.2. and 2.1.6.), was discarded and the portion remaining aseptically transferred into a labelled petri dish containing 3% malt extract agar. The petri dishes were incubated in the dark for 1 month at 25°C to allow growth of any micro-organisms present.

Growth of bacteria and moulds was recorded as such, though by reference to standard texts for identification of fungi (Barnett, 1958; Wang and Zabel, 1990) the majority of moulds were found to be strains of *Cladosporium resinae*. Suspected basidiomycetes were sub-cultured onto 3% malt extract agar, containing 4 ppm Benomyl to prevent the growth of moulds. Growth on Benomyl agar and the presence of mycelial clamp connections identified these fungi as basidiomycetes. Strains of the basidiomycete *Neolentinus lepideus*, differentiated by morphological and cultural characterisation together with cross referencing against stock cultures of this fungus, were the most commonly isolated. The identity of *N. lepideus* isolates was confirmed using Nobles (1964) identification key and their presence on wood cores recorded. The numbers of other basidiomycetes found were so low as to be insignificant.

3. RESULTS.

3.1. Tables and figures.

Tables 1 and 2 indicate initial presence, prior to remedial treatment, and final presence, after remedial treatment, of *N. lepideus*, bacteria and fungal moulds on wood cores removed from treated and untreated distribution poles respectively. At each indicated core sampling time after treatment (tables 1 and 2), separate representative groups of 20 treated and 20 untreated poles, selected on the basis of percentage moisture content, are each separated into 2 groups of 10 "dry" and 10 "wet" poles. Tables 1 and 2 also show mean pole measurements of diameter, height and depth, moisture content and percentage creosote penetration.

Figure 1 shows the original and final percentage of wood cores, from control and remedially treated poles, from which *N. lepideus*, bacteria and moulds were isolated at each period after remedial treatment. Figure 2 indicates the original and final percentage of cores recovered which were free of microbial growth, and the consequent effect on the percentage of treated and control poles from which no micro-organisms were isolated at each period after treatment.

Table 3(part A), shows the mean initial and final presence of *N. lepideus*, bacteria and moulds, combined over all sampling periods after treatment for "wet" and "dry" pole groups of remedially treated distribution poles (table 1). Similarly, table 3(part B) indicates these mean values for untreated distribution poles (table 2) and table 3(part C), displays the mean initial and final presence of these organisms on wood cores from treated and control poles irrespective of moisture status.

Oneway analysis of variance was carried out to compare the mean presence of each group of isolated organisms and clear cores (table 3(parts A, B and C)) at a number of levels. The probability values for statistically significant differences for comparisons within treated poles, within control poles and between treated and control poles are presented in tables 4, 5 and 6 respectively.

3.2. Distribution pole parameters.

Oneway analysis of variance indicated no significant differences between the mean pole parameters of initial moisture content, final moisture content, diameter, percentage creosote depth, pole depth and pole height, compared separately,

between 1, 3, 6 and 16 months sampling periods for control or treated "dry" poles, control or treated "wet" poles and control or treated poles irrespective of moisture content (tables 1 and 2). Similarly, there were no significant differences when these pole parameters were compared at 1, 3, 6 and 16 months between control and treated "dry" poles, control and treated "wet" poles and control and treated poles irrespective of moisture content. The initial mean moisture content of "dry" control and treated poles was significantly lower, at each sampling time, than the initial mean moisture content of "wet" control and treated poles respectively, ($P < 0.0005$ and 0.0005). Identical significant differences were found between final mean moisture contents for both control and treated poles (tables 1 and 2).

The procedure for selection of poles (section 2.1.5.), for removal of a 2nd set of core samples, was therefore successful in providing comparative groups of "wet" and "dry", control and treated poles.

3.3. Isolation of fungi.

3.3.1. General trends indicated by tables and figures.

Tables 1 and 2 clearly indicate that moulds made up the larger portion of isolated organisms irrespective of preservative application or sample time. The greater isolation of bacteria in poles of high moisture content from either group of control or treated poles is also shown, as is the infrequent isolation of the basidiomycete *N. lepideus*. Remedial treatment appeared to cause a decline in the presence of bacteria and moulds especially at 1 and 3 months after treatment irrespective of the moisture status of poles (table 1). Treatment also appeared to result in more wood cores from which no micro-organisms were isolated.

Figure 1, like tables 1 and 2, displays the preponderance of moulds in original and final cores from treated and control poles and again the infrequent presence of *N. lepideus*. Examination of original isolation percentages for both treated and control pole cores (figure 1) indicates the generally stable proportions of moulds, in particular, and bacteria, when cores were removed at the same date, December 1989. Comparison of the original and final isolation percentages of bacteria and moulds for control poles (figure 1) indicates a seasonal variation in pole populations of organisms. For instance, 6 months after treatment, in July 1990, the percentage of isolated organisms which were moulds rose for control poles. Conversely, isolated bacteria percentages fell at this time, which supports the indications of a preference for "wetter" conditions given in tables 1 and 2. The effects of remedial treatment appeared to be superimposed on this seasonal population variation, with

apparent reductions in percentage re-isolations of moulds and bacteria, in treated poles, up to 6 months after treatment (figure 1). The negligible isolation of *N. lepideus* within the poles studied makes meaningful comment on treatment effects difficult.

Figure 2 indicates the distinct percentage increase in cores displaying no growth for treated poles particularly at 1 and 3 months after treatment. Again, comparison between treated and control clear core percentages indicates that the treatment effect is superimposed on a seasonal population effect. The effects of remedial treatment on the percentage of poles sampled which were apparently free of microbial growth are clearly shown at 1 and 3 months after treatment (figure 2). Thereafter the treatment appears to have no beneficial effect. However, comparison of clear core and clear pole findings, e.g. final core and pole results at 16 months for treated and control poles, indicates that chance plays a large part in these findings when the number of clear cores is not excessive.

3.3.2. Isolation of *N. lepideus*.

Tables 4, 5 and 6 show the lack of significant differences for the mean presence of N. lepideus (table 3, parts A, B and C) for any comparison. However, to base a conclusion, as to the effect of Rentex treatment on N. lepideus, would be unwise given the infrequent and erratic occurrence of this organism (tables 1, 2, 3 and figure 1).

3.3.3. Isolation of bacteria.

The initial presence of bacteria was significantly higher in "wet" poles than in "dry" for both treated and control pole groups (tables 4, 5 and 3(parts A and B),) confirming the preference of bacteria for an environment of higher moisture content (section 3.3.1.). There was no significant difference in the initial presence of these organisms, between treated and control poles (tables 6 and 3(part C)) indicating the stability of populations prior to remedial treatment. The lack of significant differences between initial and final presence of bacteria in control poles (tables 5 and 3(part C)) ensured that no significant consistent seasonal effect occurred to mask the effects of the remedial treatment. The final presence of these organisms in "wet" and "dry" poles was not significantly different within treated poles (tables 4 and 3(part A)), therefore the treatment had nullified the normal population imbalance between "wet" and "dry", which still existed in control poles (tables 5 and 3(part B)). This effect of the treatment was confirmed by the significantly lower final presence of these organisms in "wet" treated poles compared with "wet"

control poles (tables 6 and 3(parts A and B)), which gave an overall significant reduction between initial and final presence of bacteria in treated poles (tables 4 and 3(part A)). Due to the treatment effect in "wet" treated poles (tables 4 and 3(part A)) and the lack of a significant seasonal effect in control poles (tables 5 and 3(part C)), the final mean presence of bacteria was significantly reduced in treated poles compared with control poles (tables 6 and 3(part C)).

3.3.4. Isolation of moulds.

The mean initial "wet" and "dry" presence respectively of moulds in treated poles was significantly lower than their mean final presence (tables 4 and 3(part A)) giving a significantly lower final mean presence overall within treated poles (tables 4 and 3(part C)). The lack of a significant difference in the final mean presence of moulds between control and treated poles (tables 6 and 3(part C)) indicated that a seasonal effect was operating. This effect did not obscure differences between the combined final mean values for moulds (table 3 part C) but it did serve to increase variation particularly for the final mean value of moulds in "wet" control poles (table 3, part B) in the absence of remedial treatment. This, combined with the significantly lower mean initial presence of moulds in "wet" control poles (tables 3 (part B) and 5.) which was reflected in the final mean presence of moulds in "wet" control poles (table 3, part B) resulted in no significant differences between the final mean presence of moulds in control and treated poles (tables 6 and 3(part C)).

3.3.5. Wood cores displaying no growth of micro-organisms.

Tables 4 and 3(part A), indicate that the final mean occurrence of clear cores was significantly greater than the initial mean occurrence at each statistical level of comparison within treated poles. The levels of significance (table 4) indicate that the treatment was more effective at higher moisture contents and this is confirmed by the significantly greater occurrence of final clear cores between "wet" treated and control poles (tables 6 and 3(parts A and B)). The mean final occurrence of clear cores in treated poles overall moisture contents was significantly greater than control pole values (tables 6 and 3(part C)). As expected, no significant differences were found within control poles (table 5). Therefore, remedial treatment did produce a consistent significant reduction in numbers of isolated micro-organisms over the 16 months of the study.

4. DISCUSSION.

Due to the small number of *N. lepideus* and other basidiomycetes isolations the remedial treatment could not be shown to have had an effect in reducing natural pole populations of basidiomycetes (sections 3.3.2.).

However, remedial treatment caused a significant reduction in the normal bacterial population of distribution poles over the 16 months of study (section 3.3.3.). The significantly lower mean final presence of moulds compared to their initial presence in treated poles in conjunction with the lack of a significant difference between these values for control poles, indicates, as for bacteria, that the treatment did have an effect in reducing mould populations (section 3.3.4.). These findings were confirmed by the significantly greater number of clear cores obtained from distribution poles following treatment (section 3.3.5.).

The absence of significant differences in the final mean presence of moulds (section 3.3.4.) between treated and control poles in no way detracts from the aforementioned indications of treatment effects but demonstrates how natural environmental variations in pole populations of micro-organisms can produce an effect in untreated poles such that any differences between treated and control poles are masked. Alternatively, environmental effects may serve to highlight a treatment effect. For instance, remedial treatment caused a significant reduction in the bacterial population of distribution poles, essentially by masking the normal significant bacterial population imbalance, in favour of "wet" distribution poles, between "wet" and "dry" poles (section 3.3.3.). Given the more efficient diffusion of fluorides in timber of higher moisture contents (Liese and Schubert, 1941; Buro and Becker, 1956; Becker, 1959; Sinclair et al; 1991), the evident preference of bacteria in this study for timber of higher moisture contents, a preference which is generally known (King, 1981), may have effectively increased the susceptibility of these micro-organisms to fluorides.

The significant effect of remedial treatment on isolations of bacteria does not indicate preservative efficacy, as bacterial decay of timber is a slow process which is not as important as decay through fungal attack, though the extensive increased porosity of wood caused by bacterial decay may facilitate entry of decay fungi (King, 1981). The real importance of bacteria in wood decay may be in terms of relationships formed between the actinomycete bacteria and decay producing organisms (King and Eggins, 1977) shown in the suppression of *N. lepideus* decay rates in Pine and Lime wood blocks due to the presence of *Streptomyces xanthochromogenus* and *S. bottrophensis* (Baecker et al, 1983), both bacteria of the

actinomycete grouping. In which case, a proven effect of preservative treatment in reducing bacterial numbers alone might be regarded as opposing efficacy.

Mould isolation reductions alone due to remedial treatment, as for bacteria, cannot be regarded as indicative of treatment efficacy, as these fungi do not cause significant strength or weight losses in timber due to their inability to degrade the cellulose and lignin, of wood cells (Butcher, 1966).

Unpublished Rentex toxicity data from this laboratory indicate that the fluoride concentration in Scots Pine sapwood required to prevent decay by strains of *N. lepideus* (BAM 20 and a pole isolate) lies between 0.03 - 0.07 %w/w and the equivalent concentrations in heartwood lie between 0.03 - 0.07 for *L. lepideus* BAM 20 and 0.07 - 0.16 for a *N. lepideus* pole isolate . These data also indicate a fluoride concentration of 0.59 %w/w is required to prevent sapwood colonisation by *C. resinae*, the mould most commonly isolated in the present study. These threshold values are not dissimilar to 0.016 - 0.064 %w/w for *L. lepideus* (Smith and Cockcroft, 1967 b) and 0.2 %w/w for decay fungi generally (Henningson and Nilsson, 1975), though the higher value of Henningson and Nilsson (1975) is identical to that quoted by Becker (1973) for fluoride resistant fungi, which *N. lepideus* is not (Richards, 1924). However, it is clear from our unpublished data and that from other sources (Becker, 1973) that *C. resinae* and moulds generally are less sensitive to fluoride than *N. lepideus*.

Therefore the significant effect of remedial treatment in reducing wood core isolations of moulds, made up largely of *C. resinae*, coupled with the Rentex toxicity data strongly indicates that pole populations of *L. lepideus* would be significantly reduced up to 16 months after Rentex treatment and hence, the incidence of internal decay in treated creosoted distribution poles would be reduced. It is also clear from the current study's data on N. lepideus that not all the decay organisms have been eradicated.

The efficacy of Rentex remedial treatment may not be maintained over the longer periods expected for original DFA groundline treatment (Smith and Cockcroft, 1967 c). A continuation of the field study detailed in Sinclair et al (1991) indicated that at 20 months after remedial treatment, fluoride concentrations in uncreosoted sapwood and heartwood were generally below accepted toxic thresholds for *N. lepideus*. This is to be expected given the failure of chromium (V1) in the formulation to "fix" diffused fluoride (Sinclair et al, 1991) leading to leaching of fluoride from Rentex treated timber in the field (Sinclair et al, 1992) and in the laboratory under simulated field conditions (Smith et al, 1993). The results presented here for the

Rentex treatment are not as encouraging as those reported by Bruce et al 1989 who found an efficient sterilizing effect of Rentex on poles artificially inoculated with N. lepideus. This difference is most likely due to the inoculated poles being more heavily treated with Rentex.

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Table 1. Initial (1) and final (2) presence of N. lepideus, bacteria and moulds on cores recovered from remedially treated field poles (standard deviations in parenthesis for means of 10).

Pole Moisture Group	Months After Treatment	Mean Pole Moisture (%)		Number of Cores (of 30 recovered) Supporting Growth of:								Mean Pole Diameter (cm)	Mean Pole Creosote (% Radius)	Mean Pole Depth (m)	Mean Pole Height (m)
				N. lepideus		Bacteria		Moulds		No Growth					
		1	2	1	2	1	2	1	2	1	2				
'DRY'	1	19.95 (2.22)	21.25 (3.71)	1	4	5	2	23	12	6	15	21.07 (2.95)	52.58 (16.00)	1.55 (0.12)	7.76 (0.62)
	3	20.15 (2.26)	20.80 (2.11)	0	0	8	1	24	8	4	21	21.20 (1.92)	55.78 (17.90)	1.61 (0.14)	8.08 (0.69)
	6	20.25 (2.12)	22.30 (2.71)	0	0	4	2	20	20	9	10	21.70 (3.05)	64.02 (13.14)	1.62 (0.20)	8.12 (0.99)
	16	20.40 (2.21)	21.90 (3.51)	4	1	4	6	19	16	8	13	22.36 (2.92)	56.01 (10.38)	1.64 (0.20)	8.22 (0.99)
'WET'	1	29.00 (6.60)	23.55 (3.95)	2	2	9	5	19	7	6	16	21.95 (2.82)	53.62 (15.47)	1.5 (0.27)	7.51 (1.37)
	3	29.55 (7.24)	29.10 (6.71)	1	0	14	4	17	7	5	19	21.92 (3.16)	57.56 (14.09)	1.55 (0.14)	7.76 (0.74)
	6	30.20 (7.36)	36.10 (12.82)	2	1	15	3	20	18	3	11	19.58 (1.28)	54.93 (18.43)	1.51 (0.12)	7.56 (0.61)
	16	30.85 (8.48)	33.95 (17.18)	2	0	10	10	22	15	2	14	21.10 (2.57)	59.37 (14.99)	1.52 (0.13)	7.61 (0.64)

Table 2. Initial (1) and final (2) presence of N. lepideus, bacteria and moulds on cores recovered from untreated field poles (standard deviations in parenthesis for means of 10).

Pole Moisture Group	Months After Treatment	Mean Pole Moisture (%)		Number of Cores (of 30 recovered) Supporting Growth of:								Mean Pole Diameter (cm)	Mean Pole Creosote (% Radius)	Mean Pole Depth (m)	Mean Pole Height (m)
				N.lepideus		Bacteria		Moulds		No Growth					
		1	2	1	2	1	2	1	2	1	2				
'DRY'	1	20.25 (3.69)	20.55 (2.34)	1	0	7	5	23	24	4	5	22.32 (2.58)	48.75 (15.58)	1.62 (0.16)	8.12 (0.80)
	3	20.65 (2.89)	21.05 (1.95)	2	3	4	6	24	18	6	11	21.01 (3.42)	61.02 (16.89)	1.53 (0.11)	7.66 (0.54)
	6	20.80 (2.85)	21.10 (2.38)	2	2	7	0	24	26	5	4	22.14 (2.43)	51.84 (13.08)	1.58 (0.20)	7.87 (0.97)
	16	21.15 (2.65)	20.00 (2.36)	2	2	6	12	20	15	9	12	23.00 (1.50)	58.44 (16.74)	1.56 (0.18)	7.81 (0.91)
'WET'	1	31.75 (7.93)	26.50 (4.02)	0	0	13	15	19	16	6	7	21.48 (1.95)	53.81 (13.96)	1.54 (0.13)	7.72 (0.67)
	3	32.25 (8.64)	26.80 (3.61)	0	0	12	14	15	8	6	11	20.63 (1.74)	60.32 (17.96)	1.5 (0.14)	7.51 (0.71)
	6	32.80 (8.60)	27.65 (6.84)	0	2	12	3	18	27	4	3	20.29 (1.40)	56.57 (11.16)	1.51 (0.12)	7.56 (0.61)
	16	33.30 (9.77)	33.60 (17.86)	5	3	14	15	16	13	4	8	20.17 (1.95)	57.08 (8.71)	1.46 (0.09)	7.32 (0.45)

Table 3. Mean initial (1) and final (2) presence of N. lepideus, bacteria and moulds, on wood cores, combined over all sampling periods from:

A - Remedially treated poles of high and low moisture content

B - Control poles of high and low moisture content

C - Remedially treated and control poles

(Standard deviations in parenthesis are for means of 4 (A, B) and 8 (C)].

	Pole Group	Moisture Status	Mean Number of Cores Supporting Growth of:							
			N. lepideus		Bacteria		Mould		No Growth	
			1	2	1	2	1	2	1	2
A	Treated	'Dry'	1.25 (1.89)	1.25 (1.89)	5.25 (1.89)	2.75 (2.22)	21.50 (2.37)	14.00 (5.16)	6.75 (2.22)	14.75 (4.65)
	Treated	'Wet'	1.75 (0.51)	0.75 (0.96)	12.00 (2.94)	5.50 (3.12)	19.50 (2.07)	11.75 (5.61)	4.00 (0.99)	15.00 (3.36)
B	Control	'Dry'	1.75 (0.51)	1.75 (1.26)	6.00 (1.41)	5.75 (4.92)	22.75 (1.89)	20.75 (5.13)	6.00 (2.16)	8.00 (4.08)
	Control	'Wet'	1.25 (2.49)	1.25 (1.50)	12.75 (0.96)	11.75 (5.85)	17.00 (1.83)	16.00 (8.04)	5.00 (1.14)	7.25 (3.30)
C	Treated	D/W	1.50 (1.31)	1.00 (1.41)	8.62 (4.27)	4.12 (2.90)	20.50 (2.33)	12.88 (5.14)	5.38 (2.39)	14.88 (3.76)
	Control	D/W	1.50 (1.69)	1.50 (1.31)	9.38 (3.78)	8.75 (5.95)	19.88 (3.52)	18.38 (6.74)	5.50 (1.69)	7.62 (3.46)

Table 4. Significance table for statistical comparisons within treated poles.

Comparison (oneway analysis of variance)	N. lepideus	Bacteria	Moulds	Clear Cores
'Wet' x 'dry' initial presence	x	0.008	x	x
'Wet' x 'dry' final presence	x	x	x	x
'Wet' initial x final presence	x	0.023	0.041	0.001
'Dry' initial x final presence	x	x	0.039	0.021
Initial x final presence	x	0.027	0.002	0.0005

Table 5. Significance table for statistical comparisons within control poles.

Comparison (oneway analysis of variance)	N. lepideus	Bacteria	Moulds	Clear Cores
'Wet' x 'dry' initial presence	x	0.0005	0.005	x
'Wet' x 'dry' final presence	x	0.015	x	x
'Wet' initial x final presence	x	x	x	x
'Dry' initial x final presence	x	x	x	x
Initial x final presence	x	x	x	x

Table 6. Significance table for statistical comparisons between treated and. control poles.

Comparison (oneway analysis of variance)	N. lepideus	Bacteria	Moulds	Clear Cores
'Wet' initial presence, T x C	x	x	x	x
'Dry' initial presence, T x C	x	x	x	x
'Dry' final presence, T x C	x	x	x	x
'Wet' final presence, T x C	x	0.002	x	0.017
Initial presence, T x C	x	x	x	x
Final presence, T x C	x	0.020	x	0.001

X = No significant difference.

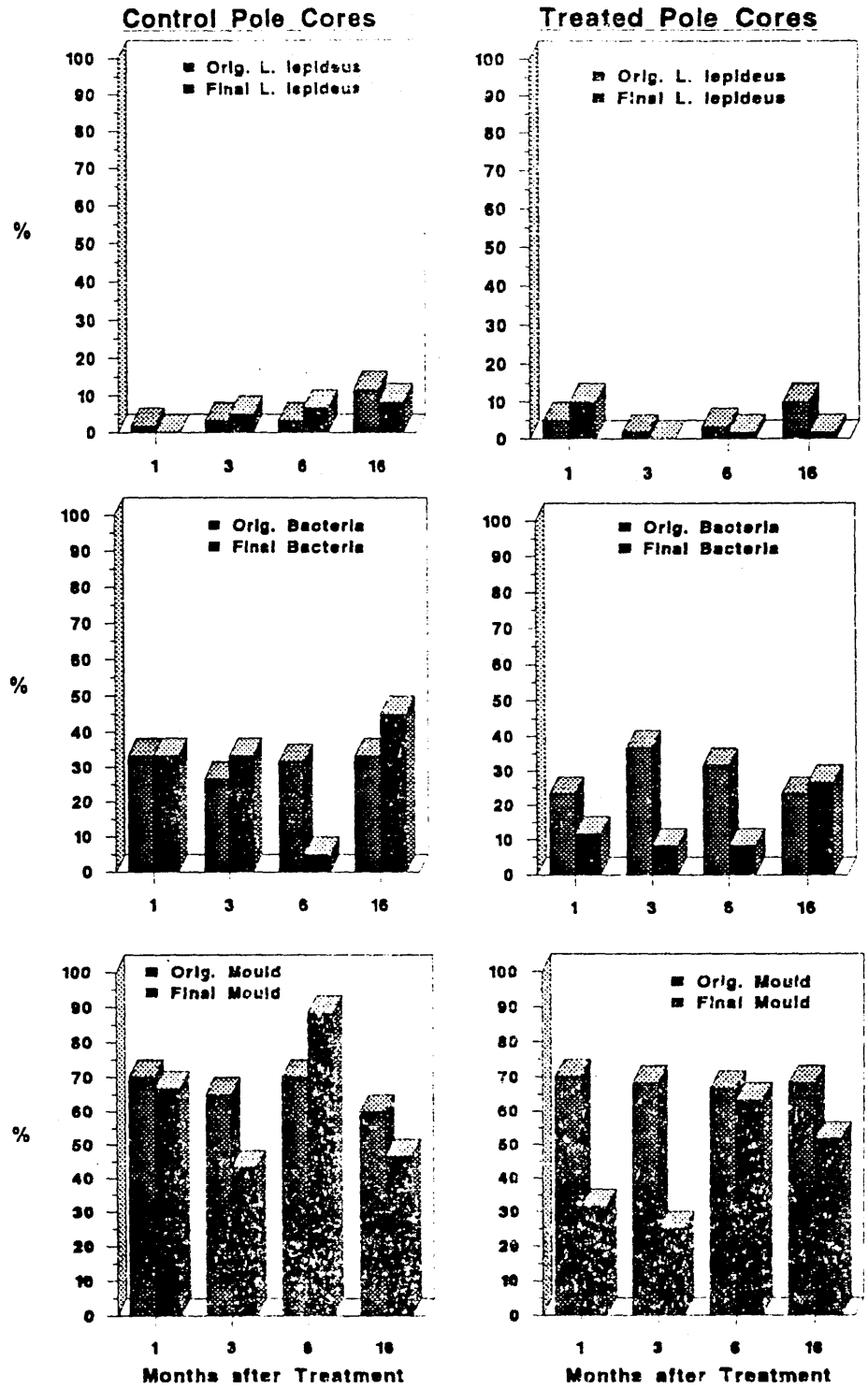

Figure 1. The number of wood cores from control and remedially treated distribution poles from which Lentinus lepideus, bacteria and moulds were isolated, expressed as a percentage of the total number of wood cores recovered. Original percentages are for cores removed, at the same time, from 4 groups of 20 poles prior to treatment and final values are for cores removed from each group of poles in succession at 1, 3, 6 and 16 months after remedial treatment.

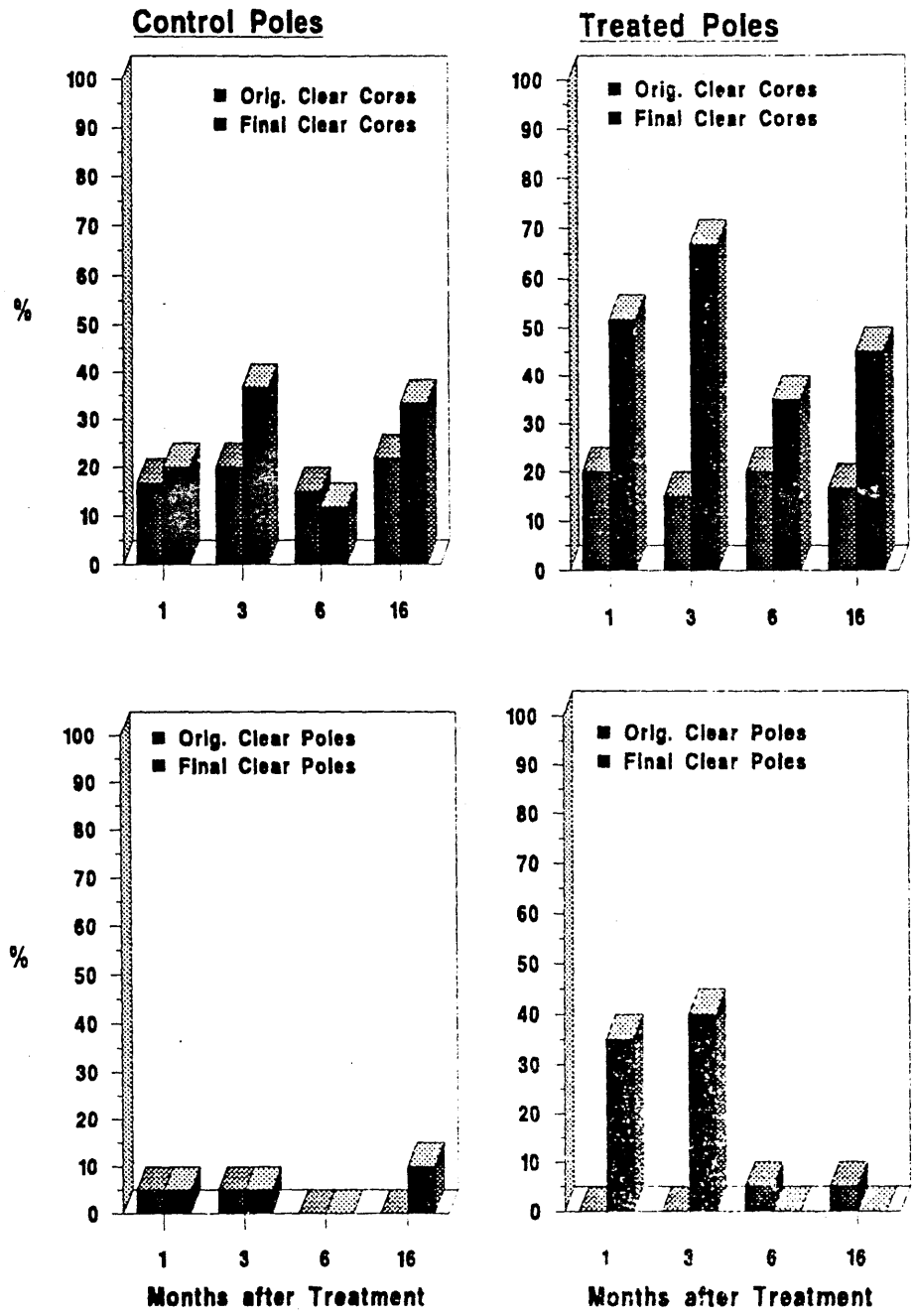

Figure 2. The number of wood cores from control and remedially treated distribution poles which were clear of microbial growth and the resultant effect on the number of poles which were clear of microbial growth, expressed as a percentage of the total number of wood cores recovered and the total number of poles sampled respectively. Original percentages are for cores removed, at the same time, from 4 groups of 20 poles prior to treatment and final values are for cores removed from each group of poles in succession at 1, 3, 6 and 16 months after remedial treatment.

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The use of a physical field model to study the effects of remedially treated timber on the growth of perennial ryegrass (*Lolium perenne*) and rye (*Secale cereale*), and the accumulation of toxic preservative constituents in *L perenne*

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The Use of a Physical Field Model to Study the Effects of Remedially Treated Timber on the Growth of Perennial Ryegrass (*Lolium perenne*) and Rye (*Secale cereale*), and the Accumulation of Toxic Preservative Constituents in *L. perenne*.

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ABSTRACT.

Low cost laboratory model systems can be used to give relatively rapid indications of the environmental effects of preservative treated timber in soil contact. This paper details the effects of remedially treated timber on the growth of crops of ***L. perenne*** and ***S. cereale*** seeded on soilbeds of different texture in close proximity to remedially treated creosoted pole sections. ***L. perenne*** sward samples were subjected to chemical analysis for fluoride and chromium content to identify bio-accumulation of these toxic preservative components. Significant but variable reductions in the dry weight yields of ***L. perenne*** samples were found in conjunction with increased fluoride and chromium contents, together with significant reductions in the density of ***L. perenne*** swards. The difference in soil texture however was found to be a more significant influence on the growth of ***S. cereale*** than the presence of treated timber. ***S. cereale*** plants within a crop canopy established on the lighter textured soil were characteristically larger and more numerous than plants from canopies on heavier soils. Results are discussed as part of an overall environmental assessment of the remedial treatment and in terms of the suitability of the physical field model as a testbed for such plant studies.

Keywords: Rentex; Physical field model; Fluoride; Chromium; Perennial ryegrass (***L. perenne***); Rye (***S. cereale***); Environmental impact.

1. Introduction.

The development of a physical field model to study indicators of the environmental impact of Rentex, a chromated fluoride wood preservative for remedial treatment of creosoted distribution poles, has been described (Sinclair et al, 1992). Three model units were constructed, each consisting of a sloping and profiled soil bed containing simulated field drains for leachate collection. Rentex treated creosoted pole sections

were positioned in two of the model units: TSS, containing sandy loam soil amended with washed sand and TS which contained the sandy loam soil only. A creosoted pole section which had not been remedially treated was positioned in control model unit, CS, which contained a sandy loam soil identical to that in model unit TS. An overhead tapwater misting apparatus provided simulated rainfall equivalent to half the annual rainfall at a field site used regularly for Rentex efficacy trials, and this was applied at intervals over a period of 40 days to the entire surface of each model unit. Lighting for each model unit was provided on a day/night cycle.

Chemical analysis of drainage waters collected from the model units during the simulated rainfall applications indicated that fluoride and chromium preservative components were leached from remedially treated timbers. However, the total quantities of fluoride found in the leach waters collected from each of the model units TSS and TS was not substantially different from that found in the leachates from CS. Also, although the total quantity of chromium in leach waters collected from model units TSS and TS was in excess of that in leachates from CS, for each of the former model units this amounted to approximately 50 mg in more than 300 litres of drainage water. Hence, the fluoride and chromium concentrations in drainage waters adjacent to treated timber were not of an order likely to cause harmful contamination of groundwater supplies (Sinclair et al, 1993).

Chemical analysis of soil samples removed from the soilbed of each model unit approximately 4 months after the final simulated rainfall application showed that fluoride and chromium concentrations adjacent to remedially treated timbers were significantly greater than background levels (Smith et al, 1993; Unpublished findings). Therefore, adsorption by soil was a major factor limiting the movement of the leached toxic preservative components in soil drainage waters. However, increased concentrations of fluoride and chromium in the surface soil adjacent to Rentex treated timber were found to have a significant negative effect on soil microbial activity . This effect was larger on the lighter textured TSS soil which was characterised by a lower organic matter content (Smith et al, 1993).

To examine any further effects and/or bio-accumulation of the leached preservative elements on plants, selected varieties were seeded to and harvested from each soilbed throughout the exposure period of these studies. Plants are particularly relevant as bio-indicators of any environmental impact of Rentex treated timber on the food chain due to their common presence around treated field poles and their known sensitivity to increased environmental concentrations of fluoride (Hansen et al, 1958; Mclaughlin and Barnes, 1975; Singh et al, 1979 a, b; Parry et al, 1984) and chromium (Hewitt, 1953; Hunter and Vergnano, 1953; Turner and Rust, 1971; Breeze, 1973; Anon, 1976; Skeffington et al, 1976; McGrath, 1982). Accordingly three crops of plants were grown consecutively, harvested, and examined in each model system.

Perennial ryegrass (***Lolium perenne***) was chosen as the primary crop species for the inclusion in the model units due to its great economic importance as the principal grass variety in grasslands which account for up to 75 % of the more intensive agricultural land in the United Kingdom (Holmes, 1980) and which predominated around remedially treated poles used in field studies of remedial treatment efficacy (Sinclair et al, 1994). Seed of the ryegrass variety '*Fennema*' was recommended by and obtained from Twyford Seeds Limited for its characteristics of disease resistance, good ground cover and persistance under wet upland conditions. This last point was of particular importance as 2 swards of grass were grown consecutively in each model unit during the period of simulated rainfall, the 1st rainfall application taking place 5 days after the 1st grass sward was sown and the final application taking place 3 days before the 2nd sward was harvested (see section 2.1). The secondary crop chosen for seeding after the two grass swards was Rye (***Secale cereale***), also supplied by Twyford Seeds Ltd., and was preferred to barley due to the formers faster growth and deeper rooting habit.

2. Materials and Methods.

2.1. Seeding and sampling of perennial ryegrass.

Ten days after Rentex treated and control pole sections were positioned in each soil bed, seed of the perennial ryegrass variety was hand sown to each soil surface at a heavy seeding rate of $90g/m^2$ to ensure a uniform growth of grass in each model unit and eliminate plant density as an experimental variable. Approximately 2 weeks later each sward was sampled. Sward samples, each covering an area 25 cm^2, were cut to a height of 2.5 cm immediately downslope of each pole section according to a sampling plan (figure 1) using stainless steel dissecting scissors and an open ended, square sided aluminium sampling tool with sides 2.5 cm deep and 5 cm long. For each soil bed, the plant material recovered from similarly lettered sample positions, A to L (figure 1), was combined and retained in closed paper bags. The samples were removed from the bags and washed in distilled water, drained, then placed in labelled clean paper bags and stored for 1 week at an ambient temperature of approximately 21°C to reach air dry weight. Chemical analysis for fluoride and total chromium content was carried out according to a modification of the method of Sinclair et al (1991). All dry weight, fluoride and chromium values reported here are expressed on an oven dry basis at 105°C .

Prior to the second grass seeding a rigid mesh of galvanised steel, consisting of 25 cm^2 sectors each containing a further 16 square sectors, was applied to the surface of each soil bed to form a fixed sampling grid. This reduced sward disturbance during

sampling while retaining the original sampling plan (figure 1). The mesh was supported 2.5 cm above the soil surface to maintain the original cutting height. Sowing of the second grass sward was carried out at the same seeding rate as the first. Approximately 2 weeks later, as this sward was cut, the number of grass leaves growing through each of the 16 square sectors within each grid section labelled 1 to 12, in each soilbed (figure 1), was recorded. For each soilbed, bulking of cut samples from grid sections A to L (figure 1), sampling of grass around the pole section and the treatment of all samples recovered was similar to that for the first grass sward.

2.2. Seeding, sampling and measurement of rye plants.

Approximately 7 weeks after the second grass sward was sampled, rye seed was sown in each soilbed at a recommended seeding rate of 645/m^2 or 383 seeds per soilbed. The seeds were planted in staggered rows at a depth of 2.5 cm.(figure 2). The first seed was planted centrally 1 cm downslope of each pole section. The distance between seed rows and between planted seeds of each row was 4 cm. Due to the reclining habit of this rye variety it was found necessary to support each plant with a short length of 2 mm glass bore tubing and plastic coated wire to ensure an erect growth habit. Approximately 3 weeks after sowing a count of seedlings was carried out to determine the number of viable plants reaching this seedling stage in each soilbed. The count was restricted to 4 seed rows either side of each pole section, numbered 1-4, and 14 rows immediately downslope of each pole section, numbered 5-18 (figure 2).

Sampling of individual plants from specific soilbed sectors, each measuring 60 cm x 10 cm downslope of each pole section (figure 3) was carried out two months after seeding. The height of each plant within the canopy and its position relative to the pole section were recorded. Each shoot was cut at the groundline, lifted free from the surrounding plants and a root marker applied. The number and total length of viable and non-viable leaves was immediately recorded for each plant. Viable and non-viable leaves were those displaying less than or more than 50% chlorosis of the leaf length respectively. Each entire shoot was retained in a paper sample bag. When all the shoots had been recovered each plants root system was gently uplifted. Adherent soil was shaken off and the roots rinsed with distilled water. There were no apparent anatomical differences between root systems recovered from different soilbeds. The length of the longest root axis was recorded for each root system and each was retained in a paper sample bag. All bags containing shoot or root samples were placed overnight in an oven set at 105°C, and the dry weight of each shoot and each root system was recorded separately.

3. Results and discussion.

Statistical comparisons of all plant measurements were carried out using oneway analysis of variance supplemented by Scheffes analysis of contrasts.

3.1. Grass swards.

3.1.1. Dry weights, fluoride and chromium contents of grass samples (table 1).

The mean dry weight yields of both first and second sward grass samples from area A/B/C of model unit CS were significantly greater than corresponding sward samples from TSS and TS, $P < \text{or} = 0.035$ (table 1). Though the yields of the first sward samples from area D/E/F of different model units were not significantly different, the yield of the second sward samples from this area of CS was significantly greater than the corresponding samples from TSS and TS, $P = 0.007$ (table 1). There were no significant differences between first or second sward grass yields from area G/H/I of different model units, and though the first sward grass yields from area J/K/L of different model units were significantly different, TS > CS > TSS at $P = 0.004$, the second yields from this sward area of different model unit were not (table 1).

These findings clearly indicated that only the dry matter yields of grass swards immediately adjacent to the remedially treated pole sections in model units TSS and TS were consistently depressed in comparison to grass swards from identical areas in model unit CS. The area and distance from the treated poles over which these yield reductions occurred in the first swards, A/B/C, equivalent to 150 cm^3 extending to 5 cm from the timber, were doubled in the second swards, areas A/B/C and D/E/F, to 300 cm^2 up to 10 cm from the timber (figure 1).

Despite these indications of reduced grass yield in sward areas A/B/C and D/E/F in close proximity to the remedially treated timbers in model units TSS and TS, there were no significant differences between the grass yields from areas, A/B/C, D/E/F, G/H/I and J/K/L of either sward in model unit TS (table 1). Similarly, the yields of the first sward samples from TSS were also not significantly different, though the yield of the second sward samples from area D/E/F was significantly greater than those from sample areas A/B/C and G/H/I of this model unit (table 1). However, within model unit CS, the yield of the first sward grass samples from A/B/C was significantly greater than that from areas D/E/F, G/H/I and J/K/L $P = 0.008$, while the yields of second sward samples from areas A/B/C and D/E/F of this model unit were significantly greater than those from G/H/I and J/K/L, $P = 0.002$ (table 1).

These findings suggested that the progressive reductions in grass yields adjacent to Rentex treated timber in model units TSS and TS, by comparison with identical areas

of grass sward in CS, was not due to a simple reduction in the normal growth of grass in the former model units. Instead, these yield reductions appeared to be caused by disruption of a progressive effect which favoured increased grass growth adjacent to the control pole section in CS. The effect of remedial treatment was to eliminate these enhanced growth conditions adjacent to the pole section. Hence, grass yields in areas immediately adjacent to the treated timbers in model units TSS and TS were generally not significantly different from yields in the rest of the sward. The favourable growth conditions found at the pole section in model unit CS may have been due to the leaching of soluble nutrients from the timber, as increased soil microbial activity was also found in the soil adjacent to the pole section in this model unit (Smith et al, 1993).

The presence of preservative fluoride and chromium as significant contaminants of the soil and its drainage waters in the soilbed area of model units TSS and TS where these grass yield reductions occurred (Sinclair et al, 1993; Smith et al, 1993), strongly implicated these preservative consituents as the causal agents. A cursory examination of table 1 indicates that the mean foliar fluoride and chromium contents of the majority of first or second sward grass samples from areas A/B/C, D/E/F and G/H/I of TSS and TS were apparently greater than those of CS. However, the variability of the individual values making up these means ensured that there were no significant differences for any inter-model or intra-model comparison. Nevertheless, the generally much higher non-significant mean foliar fluoride and chromium concentrations in grass samples from model units TSS and TS clearly shows that enhanced plant uptake of fluoride and chromium was taking place at a number of positions within these swards.

3.1.2. Sward densities of selected second sward grass samples.

Comparisons between the mean leaf numbers (table 2) in sample positions 1-6 or 7-12 (figure 1) of each model unit indicated that though significant natural variation in sward density occurred within the swards of each model unit, the mean leaf numbers in grass samples from sward positions 3 of TSS and 3 and 4 TS were significantly reduced.

Comparisons of the mean leaf numbers from corresponding sample positions from the three model units (table 2) showed few significant differences, and of these, only the mean leaf numbers in grass samples from positions 3 and 4 of TSS and TS were consistently lower than those from CS, $P < 0.0005$ and $P < 0.0005$ respectively (table 2). These findings corroborated those for intra-model comparisons in indicating that significant sward density reductions occurred over an area of 50 cm^2 extending to 5 cm from the remedially treated timber in model units TSS and TS. Given that these grass swards had been established for barely 2 weeks prior to sampling, the density

reductions close to the pole sections in model units TSS and TS probably represented reduced germination/emergence due to the presence of toxic concentrations of leached preservative fluoride and chromium.

Though sward density reductions were therefore particularly evident closest to the treated timber, the mean leaf numbers in grass samples, combined for all sample positions 1-12 of model units TSS and TS were significantly lower than that of CS, $P < 0.0005$ (table 2). Sward density was therefore significantly reduced over an area corresponding to 300 cm^3 up to 10 cm downslope of the remedially treated pole sections (figure 1). The sward density in model units TSS and TS was approximately 78 and 79 % repectively of that in CS (table 2), while dry weight yield over the same area of these model units, A/B/C and D/E/F (see table 1), amounted to approximately 72 and 69 % respectively of that in CS. This clearly indicated that the significant reductions in dry weight yield of second crop grasses in close proximity to the remedially treated pole sections (section 3.1.1.) were largely due to a fall in sward density. By implication, reduced plant size due to the toxic effects of increased foliar fluoride or chromium concentrations was of minor importance with regard to yield reductions.

3.2. Rye plant canopies.

A generalised linear ststistical model, used to examine the effect of pole section treatment, distance from the pole section, and soil type, on Rye seedling viability proportions within the model units, indicated that none of these factors had a significant effect (table 3). The differences in Rye seedling viability between model units were within the bounds of natural variation, brought about by inter-plant competition for light and nutrients within the crop. Therefore the significant increase in soil concentrations of fluoride and chromium, found by Smith et al (1993), in the region occupied by seeds planted in rows 1 - 4 in particular (figure 2), had no measureable effect on germination/emergence of this crop. Unlike, the grass swards, Rye was sown after the period of rainfall simulations, when increased concentrations of fluoride and chromium appeared in the drainage waters (Sinclair et al, 1993). Soil adsorption of these leached preservative constituents (Smith et al, 1993) probably discouraged uptake by the Rye seedlings and hence no phytotoxic effects were recorded.

Significant differences were found between different soilbed sectors within each model unit for statistical comparisons of a range of Rye crop growth measurements consisting of canopy heights and crop densities (table 4), leaf production and longevity (table 5), and dry weight yields and rooting depths (table 6). However, any negative or positive effects on the growth of Rye plants in close proximity to the pole section in each model unit were no more consistent than similar apparently random

effects which occurred throughout each crop canopy. These findings clearly indicated that any effects on the growth of Rye plants in model units TSS and TS, due to the presence of remedially treated pole sections, were no greater than those due to natural variation. This was the case even in sampling sectors within 10 cm of these timbers where concentrations of fluoride and chromium in the rhizosphere soil were greatly in excess of background levels (Smith et al, 1993). However, as the emergence/survival of Rye seedlings in this area was not affected by these soil contaminants, the lack of any negative effects in more mature plants was perhaps to be expected.

Where significant differences were found between crop growth parameters in corresponding sectors of different model units, these always favoured the plants grown in model unit TSS (tables 4, 5 and 6). Likewise, the mean leaf length, viable leaf length, shoot dry weight and root dry weight of plants over all sectors of model unit TSS were significantly greater than those of both CS and TS, at $P < 0.0005$, $P < 0.0005$, $P < 0.0005$ and $P = 0.005$ respectively (tables 5 and 6), and the mean crop canopy height in TSS was significantly greater than that found in TS, $P = 0.018$ (table 4). The crop canopy of model unit TSS also contained 30 % and 20 % more individual Rye plants than CS and TS respectively (table 4).

These findings clearly confirm the absence of a significant negative effect on Rye plant growth due to the presence of remedially treated timber, and indicate the superiority of the Rye crop grown on the sand amended soilbed of model unit TSS. The significantly greater root dry weight of plants in the soilbed of TSS (table 6) probably occurred in response to the lower water holding capacity of this sand amended soil (Sinclair et al, 1993) allied to a possibly improved soil texture for root expansion. As all the soilbeds were periodically sprayed with a standard plant fertiliser to offset possible nutrient depletion from the soil, during leaching procedures (Sinclair et al, 1993), the plants in the sand amended soilbed were therefore at a distinct advantage in terms of root surface area for nutrient uptake. Hence a better crop developed on this soil type.

5. Conclusions.

The use of the model system has been successful in allowing the simultaneous measurement and linkage of several physical and biological indicators of the environmental impact of remedially treated timber. With regard to this experiment, yield reductions in swards of Perennial ryegrass which were restricted to within 10 cm of preservative treated timber (section 3.1.1.) were predominantly accounted for by a reduction in sward density (section 3.1.2.). The sward density reductions were probably caused by a toxic effect of leached preservative fluoride and chromium at

seed germination (section 3.1.2.). As only isolated areas within the grass swards adjacent to remedially treated timbers contained foliar fluoride and chromium concentrations consistent with increased uptake from preservative contaminated soil (section 3.1.1.), the possible entry of foliar accumulations of fluoride and chromium into terrestrial food chains must be regarded as remote. Rye crops were entirely unaffected by the presence of preservative treated timber (section 3.2.) underlining the limited and short-term environmental hazard represented by the remedial treatment. The performance of this crop was primarily determined by soil type, which highlights the environmental sensitivity of the model system and its adaptability for use over a wide range of environmental studies.

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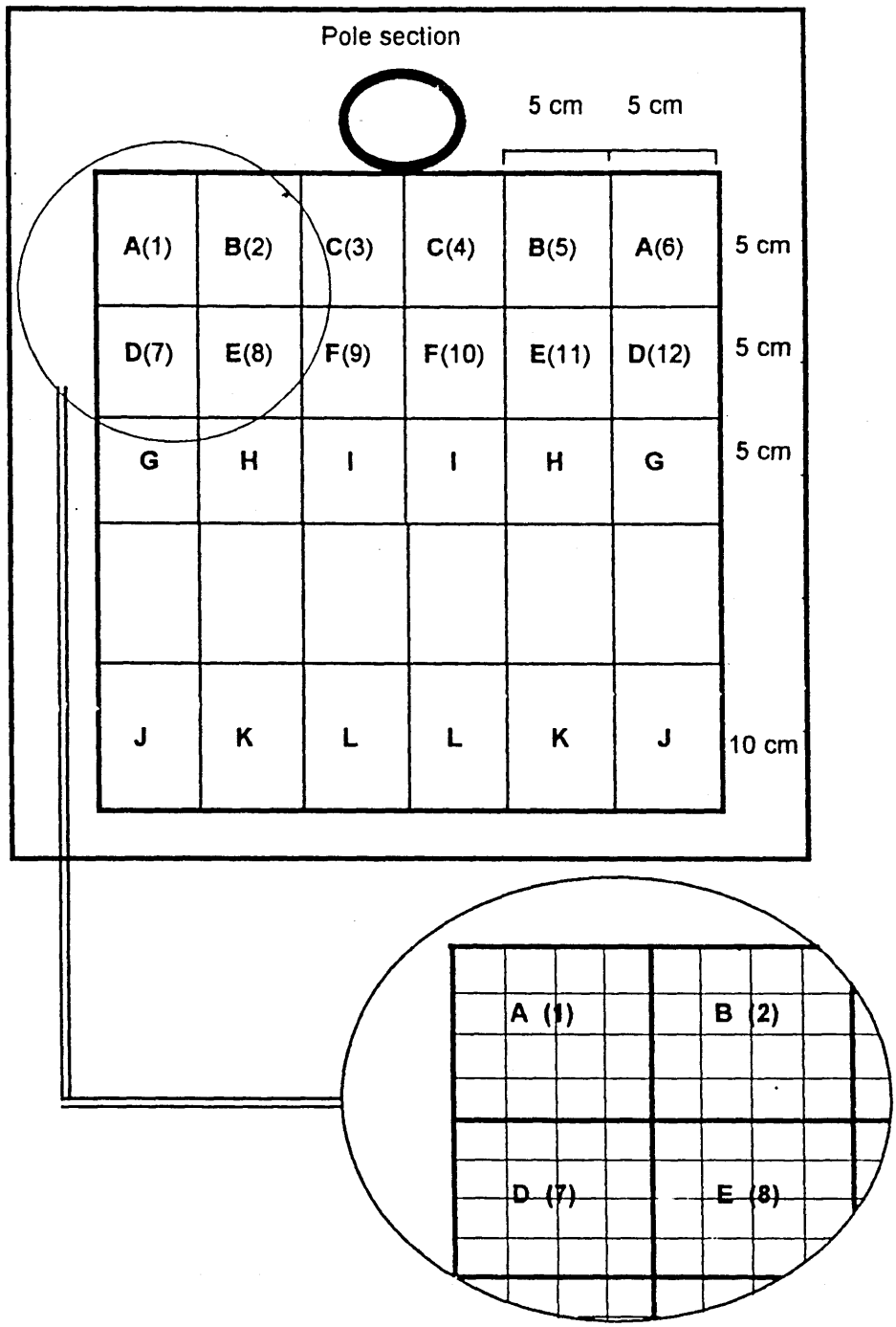

Figure 1. Diagram (not to scale) of perennial ryegrass sampling plan/grid, positioned immediately downslope of the pole section in each model unit, showing lettered and numbered sampling positions and a detail of the galvanised mesh applied to each soilbed surface prior to the 2nd sowing of ryegrass.

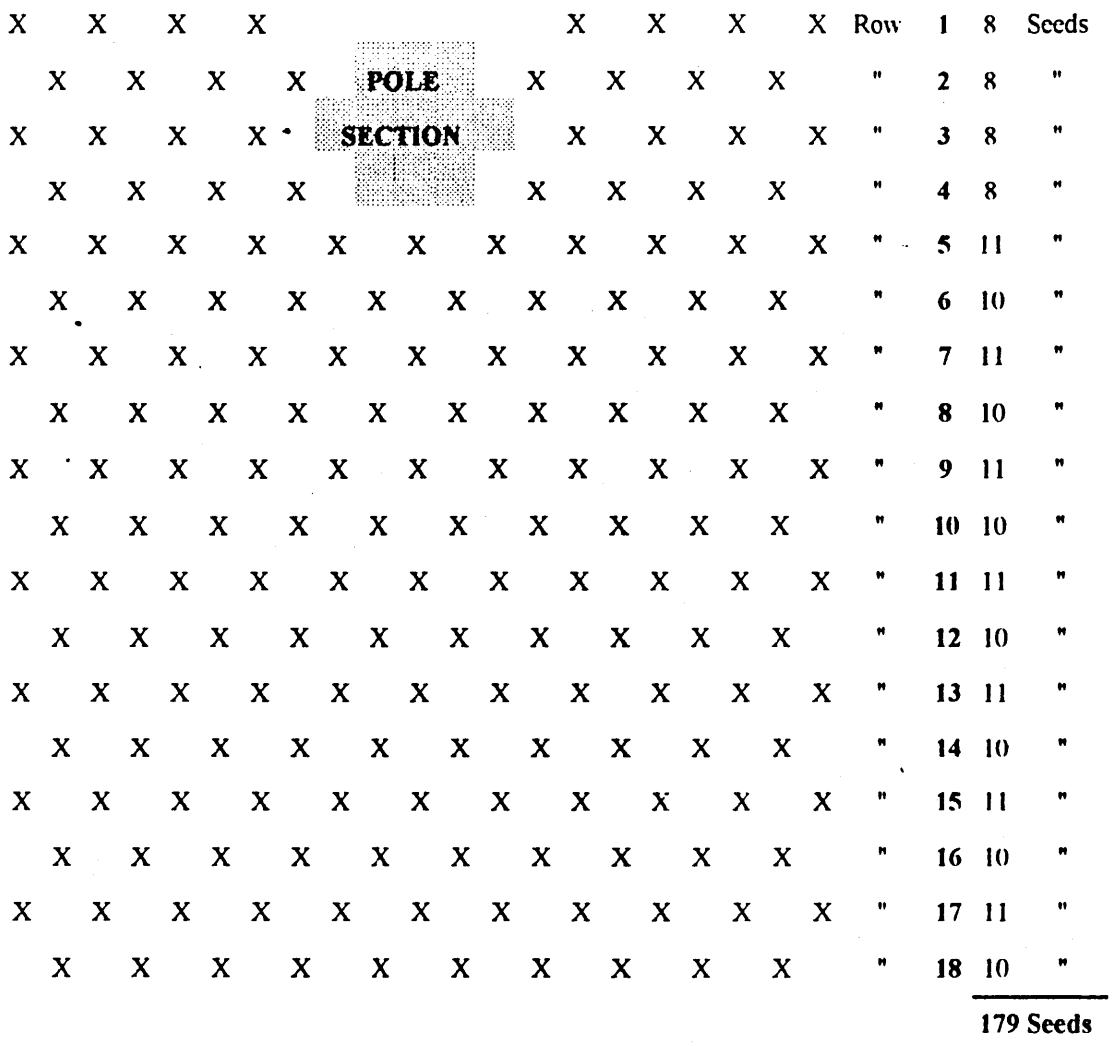

Figure 2. Plan of 18 staggered seed rows used for a count of viable Rye seedlings in each soilbed approximately 3 weeks after sowing.

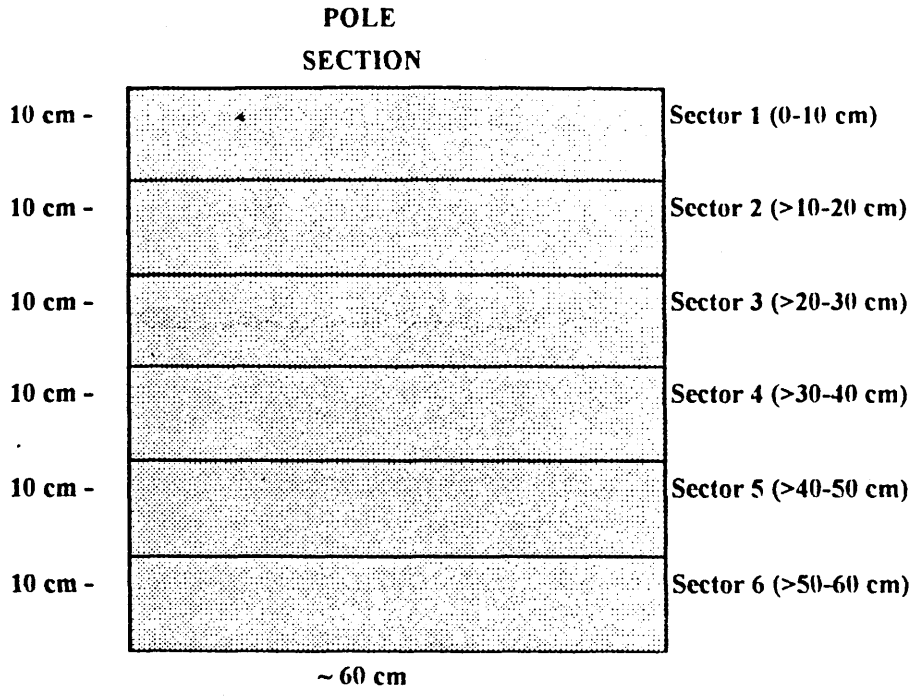

Figure 3. Diagram (not to scale) of the soilbed surface downslope of each pole section showing 6 sampling areas, of approximately 600 cm , to which individual Rye plants were assigned. This sampling plan provided a basis for statistical comparisons of Rye plants within and between model units.

Table 1. Mean dry weights (g/50 cm) and fluoride and chromium contents (ug/g) of 1st (1) and 2nd (2) sward samples combined for areas ABC, DEF, GHI and JKL (figure 1) of model units CS, TSS and TS (standard deviations in parenthesis for means of 3 All values are also expressed figuratively with I equivalent to approximately 0.02 g/50 cm for dry weight, and 4 ug/g for fluoride and chromium contents.

Parameter	Model	Area	Sward 1		Sward 2	
Mean Dry Wt. of Grass (g/50 cm)	CS	A/B/C	0.26 (0.01)	IIIIIIIII	0.21 (0.02)	IIIIIII
		D/E/F	0.20 (0.02)	IIIIIII	0.21 (0.01)	IIIIIII
		G/H/I	0.18 (0.04)	IIIIII	0.14 (0.03)	IIIII
		J/K/L	0.16 (0.01)	IIIIII	0.14 (0.01)	IIIII
	TSS	A/B/C	0.14 (0.04)	IIIII	0.13 (0.02)	IIIII
		D/E/F	0.15 (0.02)	IIIIII	0.17 (0.01)	IIIIII
		G/H/I	0.17 (0.04)	IIIIII	0.14 (0.02)	IIIII
		J/K/L	0.13 (0.01)	IIII	0.16 (0.01)	IIIIII
	TS	A/B/C	0.19 (0.04)	IIIIIII	0.14 (0.04)	IIIII
		D/E/F	0.20 (0.04)	IIIIIII	0.15 (0.02)	IIIIII
		G/H/I	0.22 (0.04)	IIIIIIII	0.14 (0.01)	IIIII
		J/K/L	0.19 (0.00)	IIIIIII	0.14 (0.03)	IIIII
Mean Fluoride Conc. (ug/g)	CS	A/B/C	04.24 (03.90)	I	06.92 (06.51)	II
		D/E/F	13.73 (01.54)	III	08.35 (07.41)	II
		G/H/I	12.79 (09.44)	III	13.87 (05.91)	III
		J/K/L	03.84 (00.70)	I	09.18 (05.05)	II
	TSS	A/B/C	51.80 (47.20)	IIIIIIIIII	25.68 (06.28)	IIIII
		D/E/F	14.85 (12.37)	IIII	19.80 (02.20)	IIIII
		G/H/I	18.27 (00.62)	IIIII	24.08 (13.37)	IIIII
		J/K/L	15.42 (07.55)	IIII	11.37 (03.97)	III
	TS	A/B/C	12.78 (3.45)	III	17.44 (13.50)	IIII
		D/E/F	10.19 (4.77)	III	06.80 (01.12)	II
		G/H/I	11.64 (6.28)	III	22.70 (19.30)	IIIIII
		J/K/L	13.93 (07.42)	III	20.20 (-)	IIIII
Mean Chromium Conc. (ug/g)	CS	A/B/C	15.80 (26.30)	IIII	04.83 (04.20)	I
		D/E/F	15.74 (15.50)	IIII	11.50 (20.00)	III
		G/H/I	08.47 (08.61)	II	05.21 (05.30)	I
		J/K/L	11.29 (10.87)	III	07.53 (09.25)	II
	TSS	A/B/C	96.10 (79.50)	IIIIIIIIIIIIIIIIII	10.57 (10.37)	III
		D/E/F	09.53 (11.28)	II	17.75 (04.31)	IIII
		G/H/I	07.53 (05.65)	II	23.10 (20.43)	IIIIII
		J/K/L	03.38 (04.79)	I	08.78 (08.79)	II
	TS	A/B/C	21.10 (19.00)	IIIII	20.38 (15.28)	IIIII
		D/E/F	16.70 (29.00)	IIII	35.10 (42.40)	IIIIIIII
		G/H/I	19.11 (16.70)	IIIII	16.55 (15.66)	IIII
		J/K/L	18.20 (08.05)	IIII	04.51 (06.38)	I

Table 2. Mean leaf numbers of 2nd cut grass samples for sample positions 1 - 12 and for all sample positions (see figure 1) of model units CS, TSS and TS (standard deviations and errors* in parenthesis for means of 16 and 192 respectively). Values are also expressed figuratively with I equivalent to approximately 1.

Sample Position	Mean Leaf Number in Model Unit: CS	TSS	TS
1	6.5 (3.25) IIIII	4.62 (3.12) IIII	11.38 (3.22) IIIIIIIII
2	8.69 (3.14) IIIIIII	7.63 (4.08) IIIIII	8.56 (5.14) IIIIIII
3	10.31 (3.28) IIIIIIII	4.62 (2.75) IIII	5.75 (3.09) IIIII
4	11.25 (3.49) IIIIIIIII	7.19 (3.25) IIIIII	3.81 (1.91) III
5	10.81 (4.49) IIIIIIII	6.81 (3.58) IIIIII	8.50 (4.62) IIIIII
6	10.56 (4.80) IIIIIIII	8.44 (4.66) IIIIII	7.94 (3.26) IIIIII
7	7.44 (3.05) IIIIII	6.00 (2.90) IIIII	8.44 (2.85) IIIIIII
8	7.25 (3.62) IIIIII	7.88 (2.73) IIIIII	6.94 (2.67) IIIIII
9	8.12 (2.87) IIIIIII	6.56 (2.56) IIIIII	7.62 (3.14) IIIIIII
10	8.25 (3.04) IIIIIII	7.5 (5.10) IIIIIII	6.38 (5.24) IIIII
11	9.69 (3.57) IIIIIIII	9.19 (3.56) IIIIIII	6.94 (2.67) IIIIII
12	12.00 (4.69) IIIIIIIII	9.69 (3.84) IIIIIIII	5.81 (3.58) IIIII
All *	9.24 (3.94) IIIIIII	7.18 (3.81) IIIIII	7.34 (3.93) IIIIII

Table 3. The number of Rye seedlings surviving to 3 weeks after seeding, in 18 rows adjacent to, and downslope of pole sections in model units CS, TSS and TS (see figure 2). Values are also expressed figuratively with I equivalent to approximately 5.

Model Unit	Seed Rows	Total Seeds Planted	No. Reaching Seedling Stage
CS	1 - 4	32 IIIII	25 IIIII
	5 - 11	74 IIIIIIIIIIIII	57 IIIIIIIII
	12 - 18	73 IIIIIIIIIIII	55 IIIIIIIII
TSS	1 - 4	32 IIIII	21 IIII
	5 - 11	74 IIIIIIIIIIII	63 IIIIIIIIII
	12 - 18	73 IIIIIIIIIIII	60 IIIIIIIII
TS	1 - 4	32 IIIII	19 IIII
	5 - 11	74 IIIIIIIIIIII	51 IIIIIIII
	12 - 18	73 IIIIIIIIIIII	57 IIIIIIII

Table 4. Mean height of tallest plant part and number of plants within sample sectors of the Rye plant canopy (see figure 3) in model units CS, TSS and TS (standard deviations and errors+ in parenthesis for means of up to 16 and 65 respectively). Values are also expressed figuratively with I equivalent to approximately 1.

Parameter	Sector	Model Unit		
		CS	TSS	TS
Mean Height of Tallest Plant Part (cm)	1	09.93 (5.09) IIIIIIII	14.96 (9.42) IIIIIIIIIII	11.98 (6.49) IIIIIIIII
	2	09.75 (4.17) IIIIIIII	12.07 (7.07) IIIIIIIII	05.53 (5.55) IIIII
	3	09.93 (5.24) IIIIIIII	15.49 (5.65) IIIIIIIIIII	07.15 (5.83) IIIIII
	4	16.53 (5.04) IIIIIIIIIIII	13.28 (6.70) IIIIIIIII	12.18 (6.75) IIIIIIIII
	5	13.00 (6.50) IIIIIIIII	13.81 (6.19) IIIIIIIIII	14.35 (5.43) IIIIIIIIII
	6	11.91 (4.06) IIIIIIIII	16.38 (4.95) IIIIIIIIIII	14.10 (6.31) IIIIIIIIII
	Mean +	11.77 (0.76) IIIIIIIII	14.26 (0.83) IIIIIIIIII	11.13 (0.91) IIIIIIII
Number of Plants	1	9 IIIIIII	11 IIIIIIII	9 IIIIIII
	2	10 IIIIIIII	10 IIIIIII	8 IIIIII
	3	8 IIIIII	11 IIIIIIII	8 IIIIII
	4	9 IIIIIII	9 IIIIIII	10 IIIIIIII
	5	6 IIIII	16 IIIIIIIIIII	8 IIIIII
	6	8 IIIIII	8 IIIIII	11 IIIIIIII
	Total	50	65	54

Table 5. Mean number of leaves, number of senesced leaves and total length of surviving leaves per plant, within sample sectors, in the Rye plant canopy (see figure 3) in model units CS, TSS and TS (standard deviations and standard errors+ in parenthesis for means of up to 16 and 65 respectively). Values are also expressed figuratively with I equivalent to 1 (A), 0.5 (B) and 10 cm (C).

Parameter	Sector	Model Unit		
		CS	TSS	TS
Mean No. of Leaves per Plant (A)	1	6.00 (0.71) IIIIII	6.36 (0.81) IIIIII	5.78 (0.44) IIIIII
	2	6.30 (0.68) IIIIII	6.90 (0.57) IIIIIII	6.00 (0.54) IIIIII
	3	6.00 (0.76) IIIIII	6.91 (0.70) IIIIIII	6.12 (0.84) IIIIII
	4	6.11 (0.60) IIIIII	6.22 (0.44) IIIIII	5.80 (0.79) IIIIII
	5	6.00 (0.63) IIIIII	6.62 (0.62) IIIIIII	5.88 (0.84) IIIIII
	6	6.38 (0.74) IIIIII	6.12 (0.84) IIIIII	5.64 (0.51) IIIIII
	Mean +	6.14 (0.09) IIIIII	6.55 (0.09) IIIIIII	5.85 (0.09) IIIIII
Mean No. of Senesced Leaves per Plant (B)	1	2.56 (0.53) IIIII	2.54 (0.81) IIIII	2.22 (0.44) IIII
	2	2.50 (0.71) IIIII	1.80 (0.63) IIII	2.38 (0.74) IIIII
	3	2.75 (0.71) IIIII	2.18 (0.75) IIII	2.62 (0.92) IIIII
	4	2.56 (0.53) IIIII	2.22 (0.44) IIII	2.00 (0.47) IIII
	5	2.00 (0.63) IIII	2.00 (0.82) IIII	2.12 (0.64) IIII
	6	2.38 (0.52) IIIII	2.12 (0.64) IIII	1.82 (0.98) IIII
	Mean +	2.48 (0.09) IIII	2.14 (0.09) IIII	2.17 (0.10) IIII
Mean Total Length of Surviving Leaves (cm) per Plant (C)	1	74.22 (15.47) IIIIIII	091.40 (36.40) IIIIIIII	87.18 (25.34) IIIIIIII
	2	82.54 (23.68) IIIIIII	126.58 (29.72) IIIIIIIIII	78.31 (21.07) IIIIIII
	3	81.39 (25.23) IIIIIII	115.90 (36.80) IIIIIIIIII	80.80 (34.00) IIIIIII
	4	82.70 (31.60) IIIIIII	095.52 (27.60) IIIIIIII	94.92 (30.42) IIIIIIII
	5	91.00 (36.90) IIIIIIII	126.57 (28.13) IIIIIIIIII	81.60 (32.70) IIIIIII
	6	86.80 (31.90) IIIIIIII	100.40 (34.70) IIIIIIII	87.56 (25.25) IIIIIIII
	Mean +	82.58 (3.75) IIIIIII	111.28 (4.25) IIIIIIIII	85.61 (3.75) IIIIIII

Table 6. Mean dry weight of shoots, length of longest root and dry weight of roots, within sample sectors, in the Rye plant canopy (see figure 3) in model units CS, TSS and TS (standard deviations and standard errors+ in parenthesis for means of up to 16 and 65 respectively). Values are also expressed figuratively with I equivalent to 5 mg (A), 0.5 cm (B) and 0.2 mg (C).

Parameter	Sector	Model Unit		
		CS	TSS	TS
Mean Dry Wt. of Shoots (mg) (A)	1	28.4 (10.2) IIIII	47.0 (17.3) IIIIIII	38.6 (10.3) IIIIIII
	2	37.4 (14.3) IIIIII	56.0 (19.9) IIIIIIII	31.6 (10.0) IIIII
	3	33.0 (10.6) IIIII	52.0 (18.6) IIIIIII	41.4 (21.6) IIIIII
	4	33.3 (12.2) IIIII	39.4 (13.4) IIIIII	40.2 (11.7) IIIIII
	5	37.2 (15.6) IIIIII	63.3 (16.8) IIIIIIIII	39.8 (11.7) IIIIII
	6	34.1 (16.5) IIIIII	44.1 (20.3) IIIIIII	37.9 (10.4) IIIIII
	Mean +	33.8 (1.8) IIIIII	51.8 (2.4) IIIIIII	38.3 (1.8) IIIIII
Mean Length of Longest Root (cm) (B)	1	3.35 (0.74) IIIIII	3.92 (1.70) IIIIII	2.50 (0.62) IIII
	2	3.90 (0.83) IIIIII	3.84 (1.03) IIIIII	3.66 (0.57) IIIIII
	3	3.66 (0.74) IIIIII	3.07 (0.60) IIIII	3.45 (0.07) IIIIII
	4	3.76 (1.01) IIIIII	4.09 (2.21) IIIIII	3.62 (0.62) IIIIII
	5	3.02 (0.93) IIIII	4.06 (1.44) IIIIII	3.57 (0.80) IIIIII
	6	2.25 (1.25) IIII	3.00 (1.40) IIIII	4.20 (0.71) IIIIII
	Mean +	3.51 (0.15) IIIIII	3.75 (0.24) IIIIII	3.50 (0.16) IIIIII
Mean Dry Wt. of Roots (mg) (C)	1	1.3 (0.8) IIIII	2.4 (1.1) IIIIIIIII	1.5 (0.4) IIIIII
	2	1.8 (0.7) IIIIIII	1.8 (0.6) IIIIII	1.1 (0.4) IIIII
	3	1.4 (0.3) IIIIII	2.5 (1.5) IIIIIIIII	1.1 (0.2) IIIII
	4	1.8 (0.8) IIIIIII	1.9 (0.9) IIIIIII	1.8 (0.5) IIIIIII
	5	1.7 (0.5) IIIIII	2.0 (1.4) IIIIIII	1.7 (0.2) IIIIII
	6	1.2 (0.1) IIIII	2.2 (1.1) IIIIIIII	1.6 (1.1) IIIIII
	Mean +	1.6 (0.1) IIIIII	2.1 (0.2) IIIIIII	1.5 (0.1) IIIIII